APPENDIX D

GROUNDWATER CONCEPTUAL MODEL

APPENDIX D

Groundwater Conceptual Model Abstract

The objective of this appendix is to describe the Hanford Site groundwater flow and contaminant transport conceptual model developed for use in the System Assessment Capability (SAC), Rev. 0.

Output from the groundwater element will support predictions of future groundwater impacts from various waste sources (past, present, and future) that will be used to allow feasibility testing of the SAC (Rev. 0) and to complete an initial assessment of risk and impact from Hanford Site waste. The data from this element feed into the Columbia River and risk and impacts technical elements (Figure D-i).

The groundwater conceptual model is an interpretation or working description of the characteristics and dynamics of the physical hydrogeologic system. The conceptual model serves to consolidate Hanford Site data (e.g., geologic, hydraulic, transport, and contaminant) into a set of assumptions and concepts that can be quantitatively evaluated. The quantitative model will be used to estimate transport of contaminants through the groundwater pathway on the Hanford Site.

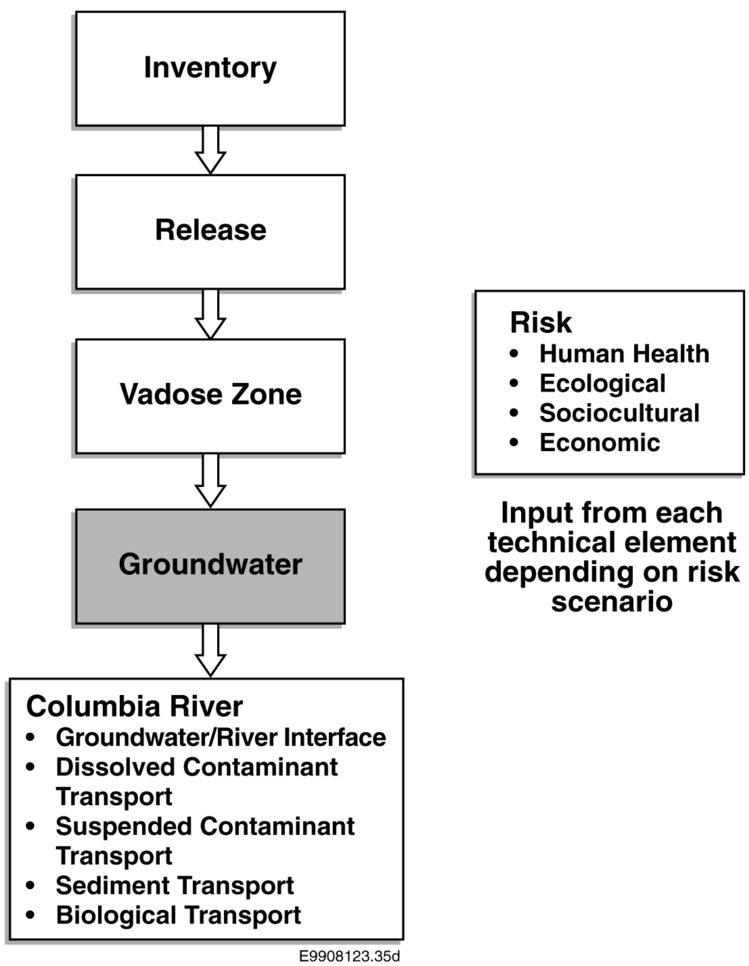
The groundwater element takes the results of the analyses from the vadose zone technical element in the form of contaminant flux from various waste sources. In addition to the influx from the vadose zone element, the groundwater model requires data that define the physical characteristics of the hydrologic system, transport parameters, and recharge and discharge rates. Definition of the hydrologic system will be based on previous subsurface investigations that have collected data on the hydrologic units, unit boundaries, hydraulic conductivity, hydraulic heads, storativity, and specific yield. Transport parameters will be based on both site-specific work done during previous investigations and on published literature values for parameters including effective porosity, dispersivity, contaminant-specific retardation coefficients, and vertical and horizontal anisotropy. Critical data to the groundwater flow and transport model also include estimates of natural recharge rates and locations and magnitude of artificial recharge to the hydrologic system, which are available from historic records and direct measurements.

Establishing model domain boundaries for the flow system will be based on site-specific knowledge and output data requirements. Boundaries will be established along the northern and eastern portion of the site corresponding to the course of the Columbia River and along the southeastern portion of the model along the course of the Yakima River. Basalt ridgelines and the Cold Creek Valley will form the western model domain boundaries. Lower flow boundaries will be established between the confined basalt aquifer system and the overlying unconfined aquifer.

The output from the groundwater element feeds the Columbia River model along the course of the river as concentrations of contaminants through time at the boundary of the groundwater/river mixing zone. The groundwater model also feeds information in the form of predicted estimates of contaminant distribution into the risk model, which will be evaluated based on selected risk metrics. Estimates of contaminant concentrations for four mobility classes of radionuclides and two chemicals will be provided as input data to the Columbia River and risk and impacts model elements.

Verification of the estimated contaminant flux through the groundwater will be made by comparing estimates of contaminant flux to the extent of contaminant distribution observed in groundwater through groundwater monitoring.

Figure D-i. System Assessment Capability System Conceptual Model.



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GROUNDWATER CONCEPTUAL MODEL

The objective of this appendix is to describe the Hanford Site groundwater flow and contaminant transport conceptual model developed for use in the System Assessment Capability (SAC), Rev. 0. The groundwater conceptual model is an interpretation or working description of the characteristics and dynamics of the physical hydrogeologic system. The groundwater model serves to consolidate Hanford Site data (e.g., geologic, hydraulic, transport, and contaminant) into a set of assumptions and concepts that can be evaluated quantitatively.

The groundwater conceptual model, which is the subject of this appendix, is the uppermost saturated zone on the Hanford Site that offers a pathway for contaminants released from past, present, and future site activities. This uppermost-saturated zone is termed the unconfined aquifer, although semi-confined conditions may exist in certain locations. The contaminant pathways may be wholly contained within the unconfined aquifer (e.g., the contaminants may be sorbed to the sediments and remain fixed at a location), or they may lead to points of discharge (i.e., along the Columbia River, to a water supply well, or to the underlying confined aquifer). Thus, the extent of the conceptual model must include both current and potential future downgradient areas from the sources to the pathway end points. The groundwater conceptual model includes the Hanford Site areas that are east and south of the Columbia River (Figure D-1). These areas lie within the Pasco Basin, a structural depression that has accumulated a relatively thick sequence of fluvial, lacustrine, and glaciofluvial sediments. Detailed summaries are provided in DOE (1988), Delaney et al. (1991), Lindsey et al. (1992), Lindsey (1995), and Cushing (1995).

The groundwater segment of the overall contaminant pathway links sources and source area with potential receptors. Radioactive and hazardous chemicals have been released on the Hanford Site from a variety of sources, including ponds, cribs, ditches, injection wells (locally referred to as reverse wells), surface spills, and tank leaks, as illustrated in Figure D-2. Many of these sources have already impacted the groundwater and some may yet cause an impact in the future. Driving forces, including natural recharge from precipitation and artificial recharge from waste disposal activities (Figure D-3), contribute to the movement of the contaminants through the vadose zone and into the groundwater of the unconfined aquifer. Several processes, including first order radioactive decay, chemical interactions with the water and sediments, and contaminant density control the fate and transport of the contaminants in the groundwater. Once in the groundwater, the contaminants move along the pathways of least resistance, moving from higher elevations to lower elevations where some contaminants may ultimately discharge into the Columbia River.

Figure D-1. The Hanford Site and Surrounding Areas.

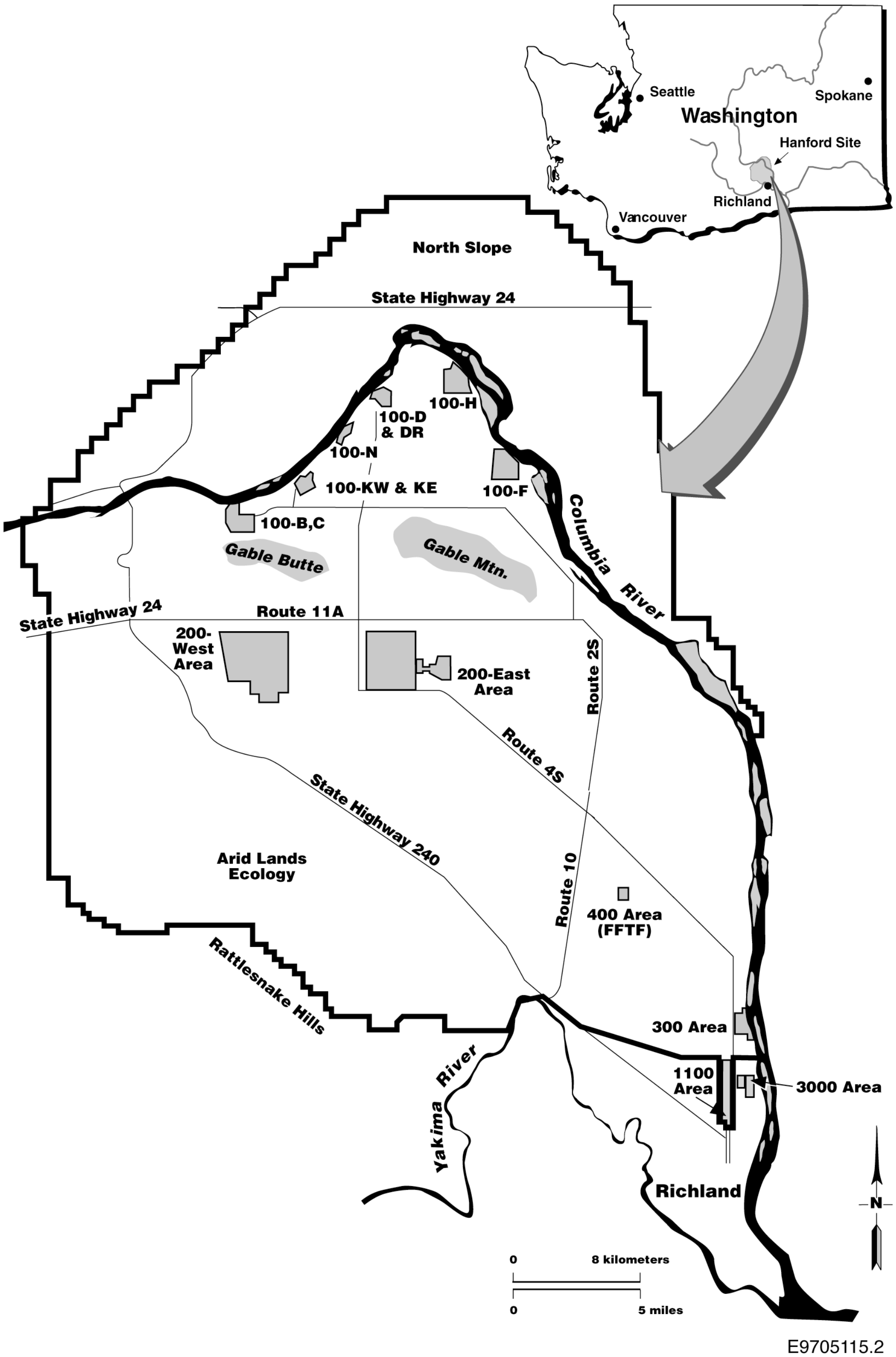


Figure D-2. Conceptualization Sources and Pathways.

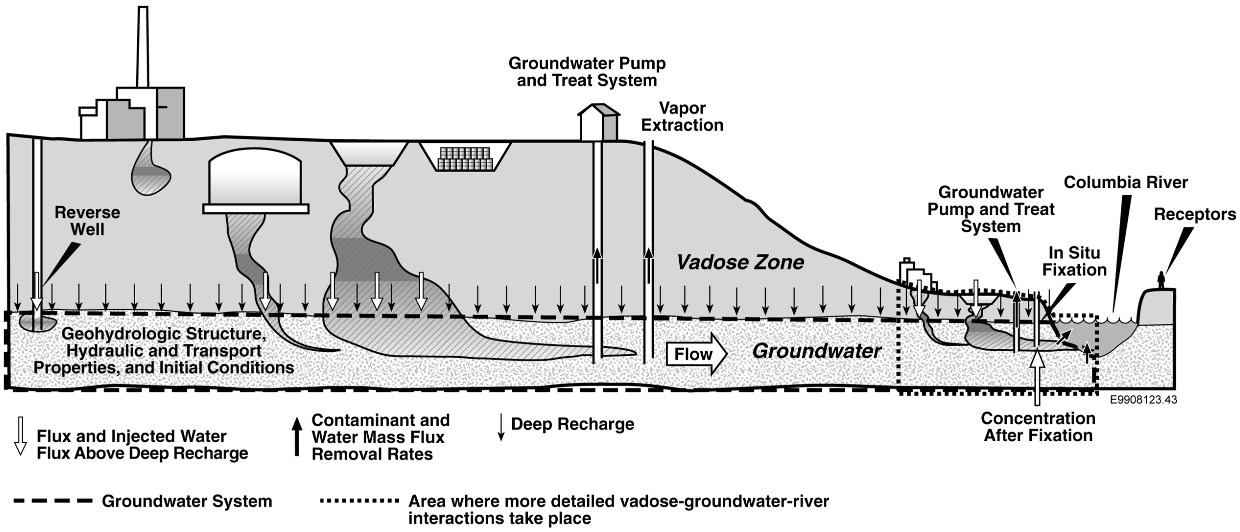


Figure D-3. Conceptual Schematic of Driving Forces, Pathways, and Transport Processes Associated with the Groundwater Conceptual Model Inputs and Outputs.

Input (spatial and temporal distributions)

- Vadose zone (contaminant and liquid mass [beyond deep recharge])
- Direct injections (reverse wells) (Contaminant and liquid mass)

Driving Forces

- Deep Recharge (natural and enhanced)
- Artificial Recharge (planned and unplanned direct discharges of liquids to ground)

Pathways

- Hydrogeologic Structure Controlled Porous Media Pathways

Fate and Transport Processes

- Isothermal, uniform density, porous media flow
- Advective-dispersive mass transport
- Transport of conservative and retarded species
- First order decay or first order reaction

Output (spatial and temporal distributions)

- River elements (mass and liquid flux and groundwater concentration entering river)
- Risk and impact assessment elements (groundwater concentration)

E9908123.44

The events and processes associated with past contamination and the present contaminant distribution in the groundwater (Figures D-4 and D-5) is illustrated in Figures D-2 and D-3. These contaminant distributions (Figures D-4 and D-5) are based on data from approximately 700 wells that are sampled at least annually to satisfy the requirements of *Resource Conservation and Recovery Act of* 1976 (RCRA), *Comprehensive Environmental Response, Compensation, and Liability Act of* 1980 (CERCLA), the Washington Administrative Code (WAC), and the U.S. Department of Energy (DOE) orders. For fiscal year 19987, the distribution of major radionuclides and hazardous chemicals in the unconfined aquifer above maximum contaminant levels (MCLs) or interim drinking water standards (DWSs) are shown in Figures D-4 and D-5, respectively.

Tritium is the most mobile and most widely distributed radionuclide contaminant in the unconfined aquifer (Figure D-4). Tritium was present in many of the Hanford Site waste streams that were discharged to the soil column and can be used to define the approximate extent of radionuclide contamination in the unconfined aquifer. In spite of the relatively short half-life of tritium (12.43 years), it was present in the groundwater in 1998 at concentrations in excess of 80,000 pCi/L, compared to the interim drinking water standard of 20,000 pCi/L. Other radionuclide contaminants found in the unconfined aquifer above MCLs or interim DWSs are strontium-90, uranium, technetium-99, and iodine-129.

Of the hazardous chemicals in the unconfined aquifer, nitrate is the most widespread (Figure D-5) and reflects its extensive use and mobility. Like tritium, nitrate can also be used to define the approximate extent of chemical contamination in the unconfined aquifer. Other hazardous chemical contaminants found in the unconfined aquifer above MCLs are chromium, carbon tetrachloride, and trichloroethylene.

This appendix provides the following: 1) a general overview of the conceptual groundwater models that have been adapted by past Hanford Site projects; 2) a description of the proposed representation of the Hanford Site groundwater conceptual model with associated assumptions, rationale, and implied uncertainties; 3) an examination of the groundwater conceptual model issues raised by the external peer review group and documented in their report (Gorelick et al. 1999); and 5) proposed path forward.

D.1 PREVIOUS AND CURRENT REPRESENTATIONS

Hanford Site geology and groundwater hydrology have been studied extensively for approximately 50 years. Since the Hanford Sites inception in the early 1940s, there have been several attempts to predict contaminant transport in the unconfined aquifer. These studies have supported several objectives, ranging from an impact assessment for the operations or remediation of a discrete facility to assessing the cumulative impacts from all Hanford waste sources. The various objectives have often resulted in differing approaches with respect to conceptualization, implementation, and spatial and temporal descretization of numerical processes. For instance, the TWRS EIS (DOE and Ecology 1996) adopted a simplified groundwater conceptual model to support an approach that used two-dimensional (2-D), steady-state groundwater flow and transient contaminant transport because a large number of long-term

Figure D-4. Distribution of Major Radionuclides in Groundwater at Concentrations Above Maximum Contaminant Levels or Interim Drinking Water Standards, Fiscal Year 1998.

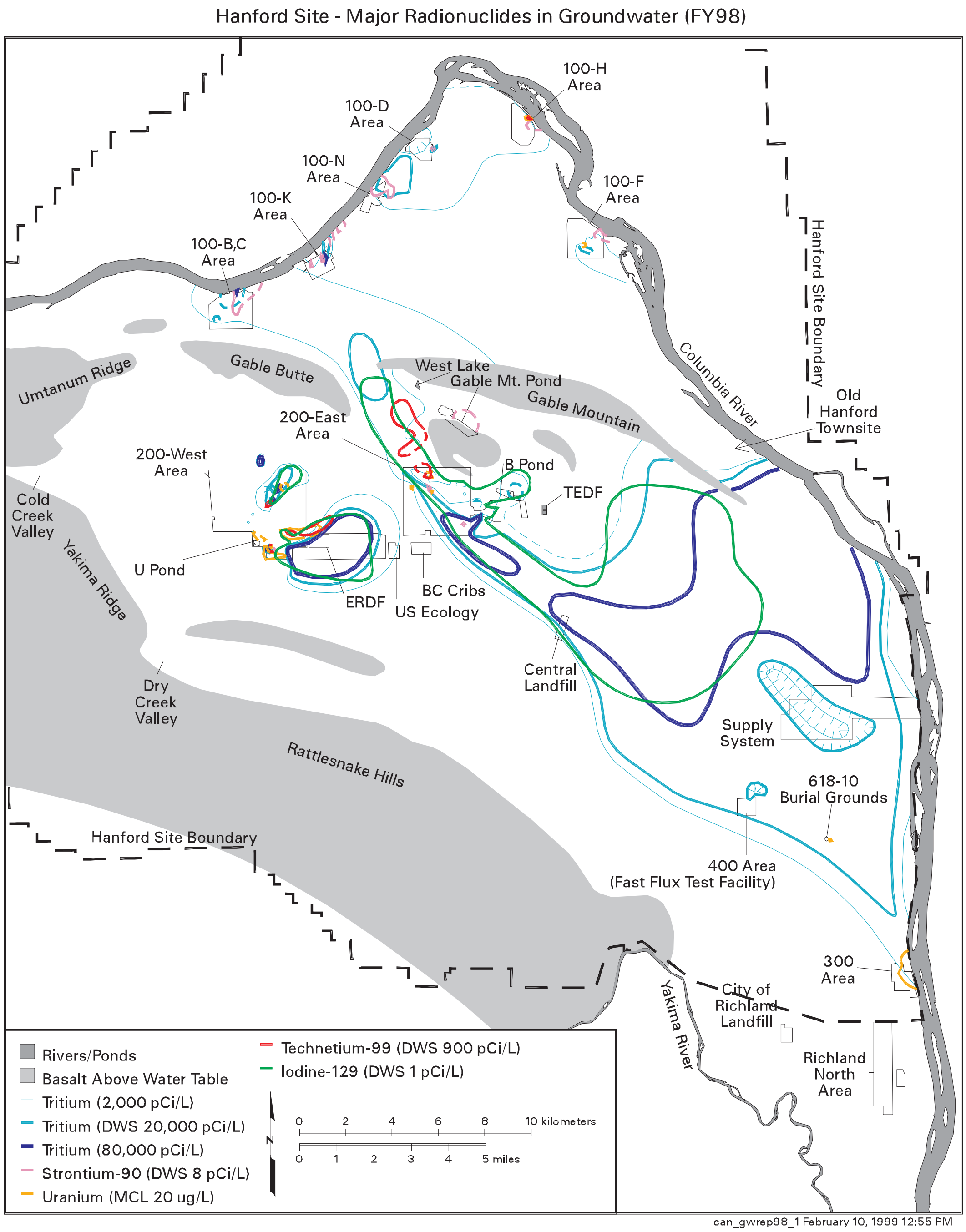
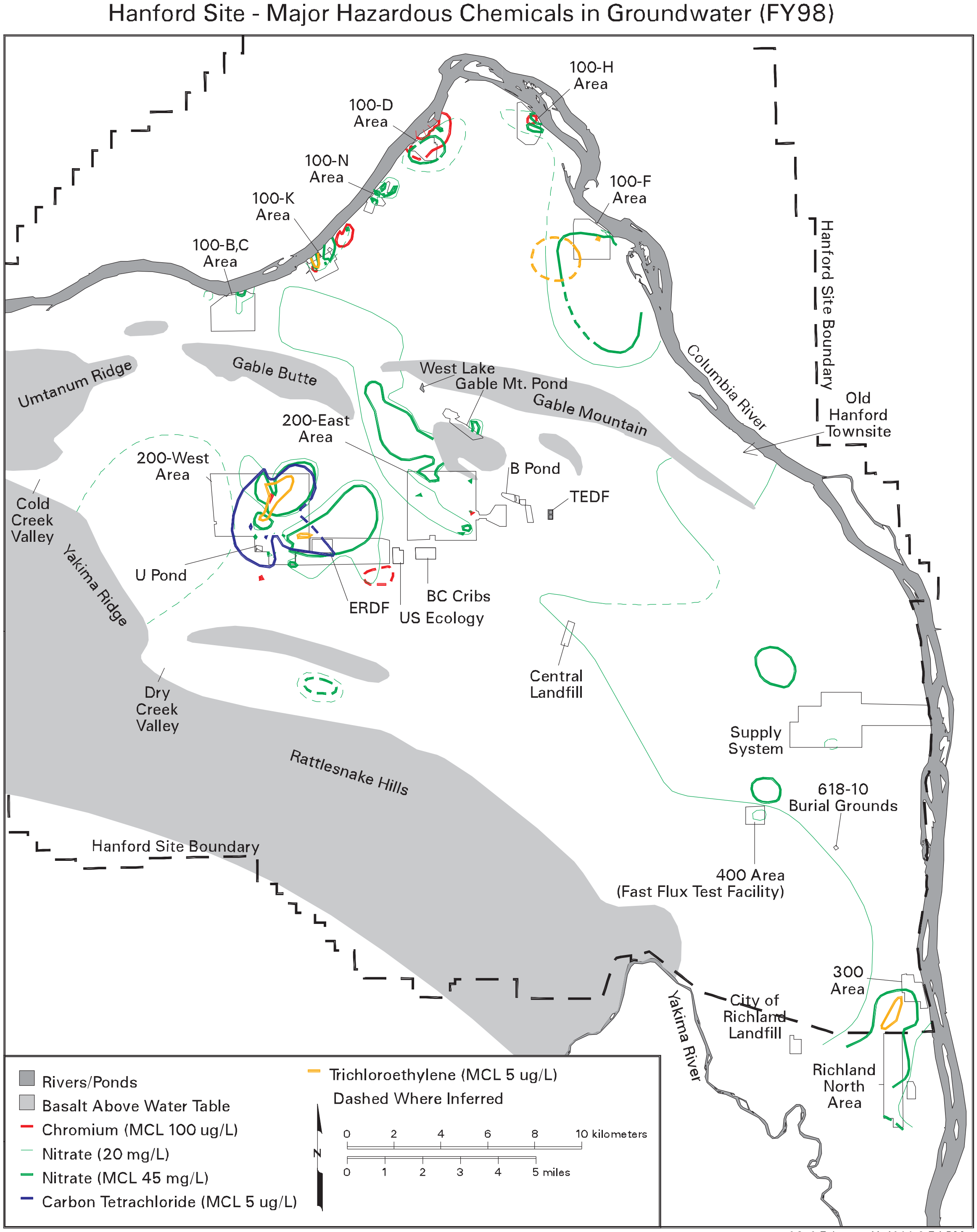


Figure D-5. Distribution of Major Hazardous Chemicals in Groundwater at Concentrations Above Maximum Contaminant Levels, Fiscal Year 1998.



scenarios (i.e., 10,000-year) had to be evaluated in a short period of time with moderate resolution. In contrast, Kincaid et al. (1998) developed a much more detailed groundwater conceptual model to support an approach that used three-dimensional (3-D) transient groundwater flow and contaminant transport because the objective was to evaluate a small number of moderate-term (i.e., 1,000-year) scenarios with sufficient resolution to distinguish between hundreds of sources. The underlying groundwater conceptual models for these various assessments have continually evolved with the addition of new data and adaptation of more powerful computers, which has lead to larger and more complex problems. The groundwater conceptual models, from these recent efforts, are described in the following subsections.

D.1.1 Objectives and Groundwater Conceptual Model Developed for the Composite Analysis

As part of the *Composite Analysis for Low-Level Waste Disposal in the 200 Area Plateau of the Hanford Site* (Composite Analysis) (Kincaid et al. 1998), site-wide groundwater modeling was performed to assess dose impacts for the offsite transport of existing plumes and future releases of contaminants in the 200 Areas (Figure D-1). Efforts were made to identify and screen all sources that could potentially interact with contaminants from Hanford Site low-level waste (LLW) disposal facilities. Inventories and projected releases of radionuclides, which are expected to contribute to the predicted doses, were established for each source. A conceptual groundwater model was developed to help predict contaminant transport through the unconfined aquifer under transient conditions, in part, because cessation of wastewater discharges at the Hanford Site during the past several years continues to cause a decline in water levels in the unconfined aquifer.

The geologic and hydrologic data used in the sitewide model which was used in the Composite Analysis, are discussed and summarized in the conceptual model report by Thorne et al. (1994) and the status report on the 3-D model implementation by Wurstner et al. (1995). As discussed in Thorne et al. (1994), the data needed to develop the 3-D conceptual model were derived from a variety of previous studies and ongoing Hanford Site investigations, as well as from work conducted specifically to support the site-wide model.

Hydraulic property data were obtained from the results of hydraulic tests documented in Bierschenk (1959), Kipp and Mud (1973), Deju (1974), Lindberg and Bond (1979), Graham et al. (1981), DOE (1988), Liikala et al. (1988), Thorne and Newcomer (1992), Peterson (1992), Connelly et al. (1992a), Connelly et al. (1992b), Swanson (1992), Thorne et al. (1993), Connelly (1994), and Swanson (1994). Information was also obtained from new tests and tests that were previously undocumented. Information on the subsurface geologic framework came primarily from interpreting geologic descriptions of samples acquired during well drilling. These interpretations were based on work by Lindsey et al. (1991), Lindsey (1992), Lindsey et al. (1992), Lindsey and Jaeger (1993), Lindberg (1993a, 1993b), Hartman and Lindsey (1993), and Swanson (1992) in the 100, 200, and 300 Areas of the Hanford Site (Figure D-1), which use the lithofacies units outlined in Lindsey (1991).

From these data and reports, the Composite Analysis adopted a hydrogeologic framework consisting of nine hydrogeologic units. These units are further discussed in the description of the proposed base conceptual groundwater model (Section 1.3.1). Hydraulic properties important to the conceptual model include both horizontal and vertical hydraulic conductivity, storativity, and specific yield for each hydrogeologic unit. Hydraulic properties used in the model are summarized in DOE (1988), and Thorne and Newcomer (1992). Thorne and Newcomer (1992) and Wurstner et al. (1995) analyzed the aquifer tests, many of which were single-well pumping tests, and selected the set of aquifer transmissivity calibration data used in the 2-D inverse model.

The boundaries of the unconfined aquifer adopted by the Composite Analysis are defined to be the Columbia River to the north and east and basalt ridges on the south and west. The Columbia River represents the regional discharge for the unconfined aquifer. The amount of groundwater discharging to the river at any location and time is a function of the local hydraulic gradient and the local aquifer properties (specifically, the hydraulic conductivity and saturated thickness). The hydraulic gradient is highly variable at any given time, because it is affected directly by the river stage, which changes on a seasonal basis in response to precipitation and temperatures within the entire Columbia River basin upstream of the Hanford Site. The river stage, and thus hydraulic gradient, are also affected by weekly and daily changes in river flows at dams on the river (most notably the Priest Rapids Dam immediately upstream of the Hanford Site), as determined by electric power generation needs, fisheries resources management, and other dam operations. More details regarding boundary conditions are described in Section 1.3.1.2.

The conceptual groundwater model for the Composite Analysis included natural recharge to the unconfined aquifer system occurring from infiltration of 1) runoff from elevated regions along the western boundary of the Hanford Site; 2) spring discharges originating from the confined basalt aquifer system; 3) precipitation falling across the Hanford Site; and 4) recharge that occurs along the Yakima River in the southern portion of the Hanford Site. Areal recharge from precipitation at the Hanford Site is highly variable, both spatially and temporally, and depends on local climate, soil type, and vegetation, as discussed in Fayer and Walters (1995). Fayer and Walters (1995) developed estimates of recharge distributions for conditions in 1992 and 1979. Both estimates illustrate the spatial variability in recharge resulting from the site-wide variation of the above-mentioned controlling parameters. Their 1979 estimate of areal recharge was used in the earlier 3-D model development efforts (Wurstner et al. 1995), as well as in the current Composite Analysis.

The other source of recharge to the unconfined aquifer is artificial recharge from wastewater disposal. During the past 50 years the large volume of wastewater discharged to disposal facilities at the Hanford Site has significantly affected groundwater flow and contaminant transport in the unconfined aquifer. The volume of artificial recharge has decreased significantly during the past 10 years and is continuing to decrease. The major discharge facilities considered in this analysis are summarized in Wurstner et al. (1995). The major wastewater discharges from both past and future sources are summarized in Cole et al. (1997).

D.1.2 Objectives and Groundwater Conceptual Model Developed for the Environmental Restoration Program

The Hanford Site-Wide Groundwater Remediation Strategy describes the approach to remediate the major groundwater contaminant plumes in the 100 and 200 Areas. As part of the strategy, a site-wide groundwater model was developed and used to estimate the effectiveness of alternative groundwater cleanup approaches (e.g., pump-treat-reinject, impermeable walls, and hydraulic control measures) to 1) support planning and implementation of remediation alternatives; 2) support risk assessments; and 3) evaluate the impact of changes in the groundwater flow field. The groundwater conceptual model developed for the Hanford Site-Wide Groundwater Remediation Strategy is described in Law et al. (1997).

The geologic and hydrogeologic conceptual models were based primarily on a synthesis of data and information presented in a number of previous studies (Law et al. 1997). The geologic model was based primarily on Lindsey (1995) with the geologic mapping taken from Reidel and Fecht (1994a, 1994b). A new map of the top of the basalt bedrock was developed for this study. The geologic mapping and the top-of-basalt surface map are part of the Hanford Environmental Information System (HEIS) database. The bottom of the unconfined aquifer system was generally taken to be the top of the basalt or the top of the lower mud unit of the Ringold Formations.

Recharge to the unconfined aquifer was assumed to occur from the Cold Creek and Dry Creek basins. The actual recharge rate used was determined during the calibration. Recharge from the surface (due to natural precipitation) and recharge from the confined aquifer was assumed to be negligible. Discharge to the Columbia River was modeled. Artificial recharge from the major liquid waste disposal facilities in the 200 East and West Areas was based on available reports (see Law et al. [1997] for values used).

Hydraulic conductivity data were used from aquifer tests reported in Connelly et al. (1992a, 1992b) and Thorne and Newcomer (1992). Two hydrostratigraphic units were represented in the conceptual model, the pre-Missoula/Hanford formation and the Ringold Formation. Hydraulic conductivity and porosity were assumed isotropic and were varied spatially in the horizontal plane. Hydraulic properties within each of the two hydrostratigraphic formations were vertically homogeneous. Vertical hydraulic conductivities were assumed to be one-tenth of the horizontal value.

D.1.3 Objectives and Groundwater Conceptual Model Developed for the Retrieval Performance Evaluation Report

In 1996 the Hanford Tanks Initiative (HTI) Project, which is jointly funded by Environmental Management (EM)-30 (TWRS) and EM-50 (Technology Deployment), selected the AX Tank Farm to demonstrate the following: 1) the methodology for establishing requirements, constraints, and criteria for retrieval technology development and deployment; 2) requirements for retrieval leakage loss during SST waste retrieval; 3) the required extent of retrieval; and 4) tank farm closure. This required assessing the Site-wide groundwater impacts from past, present, and future AX Tank Farm-related activities.

The conceptual model used for the *Retrieval Performance Evaluation Methodology for the AX Tank Farm* (RPE) DOE (1999) was modified from that used in the Composite Analysis (Kincaid et al. 1998). The Composite Analysis conceptual model adopted a geologic framework that consisted of nine layers and was designed as a 3-D transient flow and transient transport model for groundwater (Wurstner et al. 1995, Barnett et al. 1997, Cole et al. 1997). In general, the hydraulic and transport parameters adopted for the RPE site groundwater model can be traced back to the site groundwater model used in the Composite Analysis, although implementation is significantly different. The large number of strategies that required analyses, coupled with a long time period of interest (10,000 years), necessitated an approach that was computationally efficient yet provided the appropriate level of detail. Thus, the geologic framework of the RPE conceptual model was reduced to two dimensions in the horizontal plane.

D.1.4 Objectives and Groundwater Conceptual Model Developed for the TWRS EIS

The *Tank Waste Remediation System, Hanford Site, Richland, Washington, Final Environmental Impact Statement* (TWRS EIS) (DOE and Ecology 1996) was prepared by DOE and the Washington State Department of Ecology (Ecology) to fulfill *National Environmental Policy Act of 1969* (NEPA) requirements for evaluation of the retrieval alternatives being considered for application to the 177 high-level waste tanks on the Hanford Site. As part of the TWRS EIS, environmental consequence analyses were performed to evaluate the impacts of a number of tank waste management alternatives. These alternatives included continued management alternatives with no retrieval, minimal retrieval, partial retrieval, and extensive retrieval. The conceptual groundwater model adopted for the TWRS EIS was 2-D in the horizontal plane, and had an areal extent and boundary conditions similar to that of the Composite Analysis model.

The first phase of the modeling effort entailed establishing the 1979 steady-state flow field that was consistent with previous site-wide groundwater flow simulations (Wurstner and Devary 1993). This was accomplished by adopting, as closely as possible, the hydraulic parameters from the previous effort. This effort used EarthVision and ARC/INFO software capabilities to translate parameter distributions used for the CFEST (Gupta et al. 1987, Cole et al. 1988) version of the site-wide model into the TWRS EIS model.

D.2 GROUNDWATER CONCEPTUAL MODELS CONSIDERED FOR SYSTEM ASSESSMENT CAPABILITY (REV. 0)

The groundwater conceptual model will be specifically tailored to support the purposes of the initial SAC (Rev. 0). Alternative representations, in addition to the base conceptual model, are provided in this section to support consensus building between DOE, Ecology, Tribal Nations, and interested stakeholders.

As a prototype, the initial SAC will demonstrate that an assessment of the scale and scope of the Hanford Site and the Columbia River can be conducted. While the initial assessment will be limited in some respects, the assessment will be designed to:

- Examine radioactive and hazardous chemical contaminants that are expected to be the dominant contributors to risk and impacts

- Determine the long-term migration and fate of contaminants in the Hanford Site operational areas (i.e., 100, 200, and 300 Areas)

- Include a quantification of uncertainty

- Include a broad suite of quantitative and qualitative risk and impact metrics.

In addition to these general objectives, the initial assessment will be designed to distinguish the risk and impacts of the various waste types within each operational area, and sources located in the different areas (e.g., plateau sources versus near-river sources). In the SAC (Rev. 0), scenarios that pose perturbations from the baseline will not be examined. Such perturbations will be examined in later iterations. Thus, the baseline assessment will assume a static situation for many features and events. Examples are assumptions that the Columbia River will remain as it is today for the duration of the assessment, and that the climate of the region will remain unchanged. Correspondingly, assumptions of river flow and erosion and deposition patterns from wind and runoff will also remain unchanged from the current setting.

The assumption that existing conditions prevail is extended to the background contamination upon which Hanford Site contaminants are superimposed. Background contaminant levels resulting from occurrences such as fallout, mining, and agriculture, are assumed to continue, and the SAC (Rev. 0) will provide an estimate of Hanford Site contribution above these background conditions. Where total contamination levels are observed and simulated, results will be presented as the total contamination level and the Hanford Site contribution to the total.

The SAC (Rev. 0) is posed as a post-closure analysis of human and ecological health and cultural and socioeconomic impacts. In the SAC (Rev. 0) past and future disposals, remedial actions, and tank waste recovery operations are considered when they occurred or when they are planned to occur.

D.2.1 Overview of Proposed Base Groundwater Conceptual Model

The base groundwater conceptual model for the SAC (Rev. 0) is described in this section. It is referred to as the "base" groundwater conceptual model because it is believed to be reasonably comprehensive in its inclusion of hydrogeologic detail and concepts. Thus, the model forms a "base" that can be built upon and refined as additional data and information is developed. The framework of the conceptual model was developed from data on the groundwater flow system and inferences that can be drawn from the data. The geologic and hydrologic data used in the model were summarized in Wurstner et al. (1995) and are based on a number of reports published for the Hanford Site. Also provided are alternative representations of some concepts for consideration in the draft groundwater conceptual model.

Where data are available, they may be used directly. For instance, hydraulic conductivity values may be assigned to a hydrostratigraphic unit at a specific location based on an in situ test.

Where data are not available--such as at locations where no hydraulic tests were conducted (e.g., pumping test or packer test data)--correlation between other hydrostratigraphic unit characteristics (i.e., grain size distribution) have been used. Other inferences have also been used. For example, water level measurements in wells that are a part of the available data and directions of groundwater flow can be inferred from these data. The conceptual model was also developed from information on the hydrogeologic structure of the aquifer, spatial distributions of hydraulic and transport properties, aquifer boundary conditions, and the distribution and movement of contaminants.

D.2.1.1 Hydrogeologic Setting. The Hanford Site and adjacent areas north and east of the Columbia River lie within the Pasco Basin, a structural depression that has accumulated a relatively thick sequence of fluvial, lacustrine, and glaciofluvial sediments. Figure D-6 shows the surface geology and major structural features of the area. The Pasco Basin and nearby anticlines and synclines initially developed in the underlying Columbia River Basalt Group, a sequence of continental flood basalts covering more than 160,000 km^2 (DOE 1988). These basalt flows erupted as a fluid, molten lava during the late Tertiary Period. The most recent, laterally extensive basalt flow underlying the Hanford Site is the Elephant Mountain Member of the Saddle Mountains Basalt Formation, although the younger Ice Harbor Member is found in the southern part of the Hanford Site (DOE 1988). Sandwiched between various basalt flows are sedimentary interbeds collectively called the Ellensburg Formation. The Ellensburg Formation includes fluvial and lacustrine sediments consisting of mud, sand, and gravel which, along with the porous basalt flow tops and bottoms, form confined basalt aquifers across the basin. The Rattlesnake Ridge Interbed is the uppermost laterally extensive interbed and confined basalt aquifer of the Ellensburg Formation (Spane and Vermeul 1994).

Overlying the basalt within the Pasco Basin are fluvial and lacustrine sediments of the Ringold Formation (Newcomb and Strand 1953, DOE 1988, Lindsey et al. 1992). Figure D-7 shows the generalized geologic column for the Hanford Site. The ancestral Columbia River and its tributaries flowed into the Pasco Basin, depositing coarse-grained sediments in the migrating river channels and fine-grained sediments (silt and clay) in the overbank flood deposits. On at least two occasions, these river channels were blocked, forming a lake in the Pasco Basin and depositing extensive layers of fine-grained sediments within the Ringold Formation. The Plio-Pleistocene unit, consisting of a paleosol/calcrete and/or basaltic sidestream sediments, and the early “Palouse” soil, an eolian sand and silt deposit, overlie the Ringold Formation, but are present only in the western portion of the Pasco Basin. The uppermost sedimentary unit covering much of the Hanford Site is the Hanford formation, a complex series of coarse- and fine-grained sediments deposited by cataclysmic floods (called the Missoula floods) during the last ice age. For the most part, the fine-grained sediments are found near the margins of the basin and in areas protected from the main flood currents, which deposited the coarse-grained sediments. A thin veneer of eolian sands and/or recent fluvial deposits cap the Hanford formation in many areas.

Figure D-6. Surface Geology and Structural Features of the Pasco Basin.

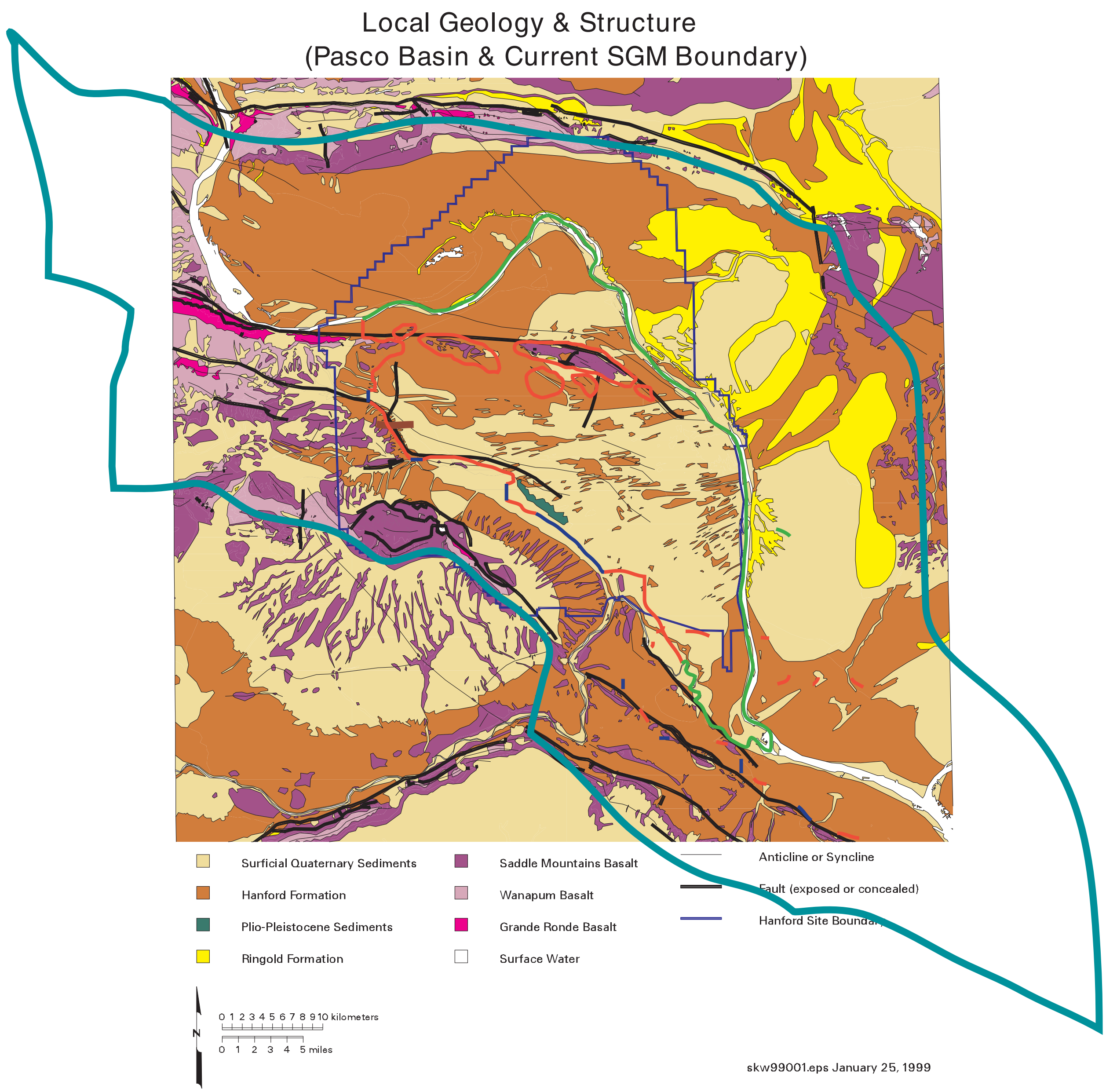
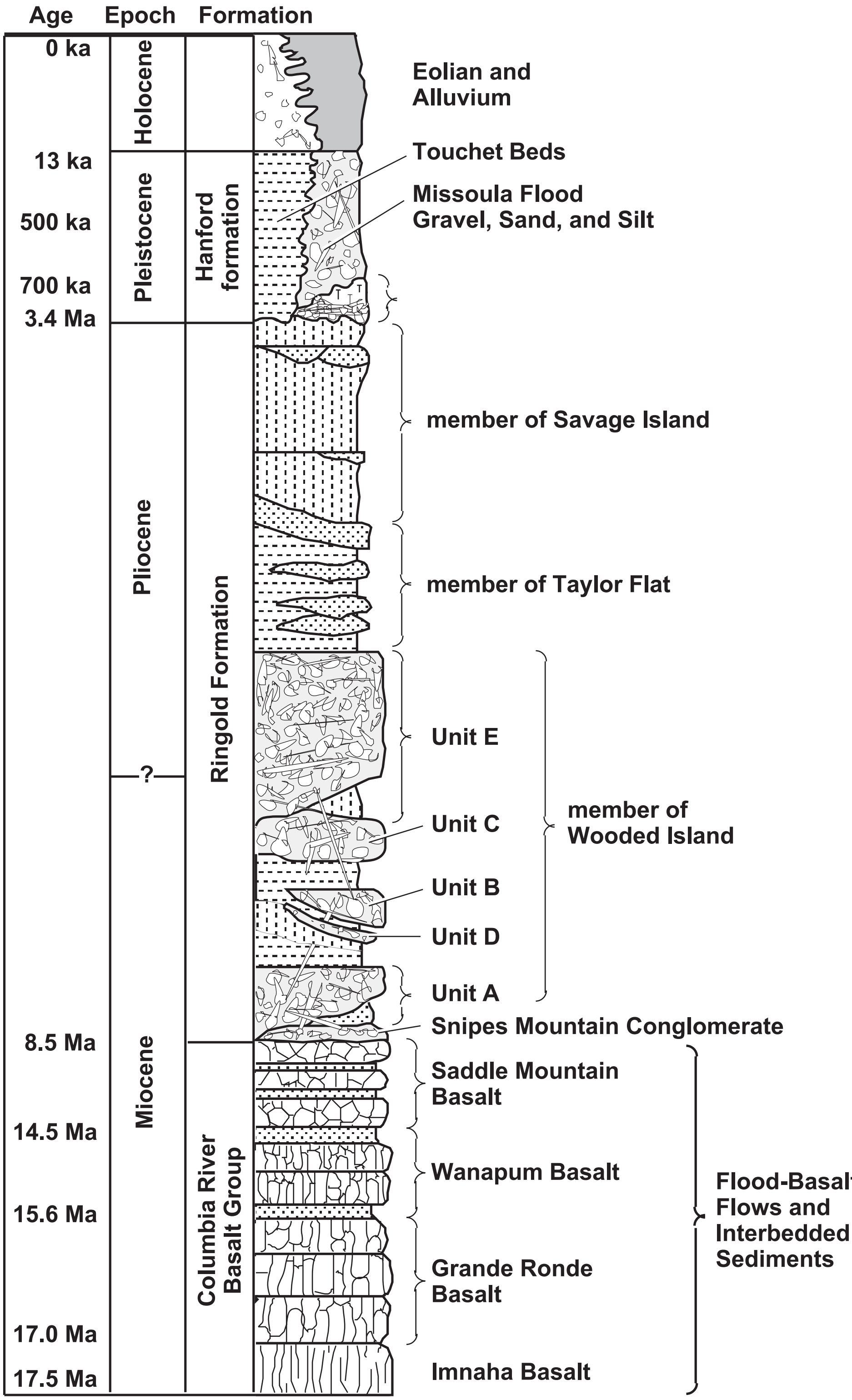


Figure D-7. Generalized Geology and Hydrogeologic Stratigraphic Columns.



The fine-grained layers of the Ringold Formation have a much lower permeability than the coarse-grained layers, forming aquitards. However, these aquitards are usually not continuous across the Hanford Site, allowing interflow between different parts of the suprabasalt aquifer on a site-wide scale. Consequently, the suprabasalt aquifer is considered one entity commonly referred to as the "Hanford unconfined aquifer system."

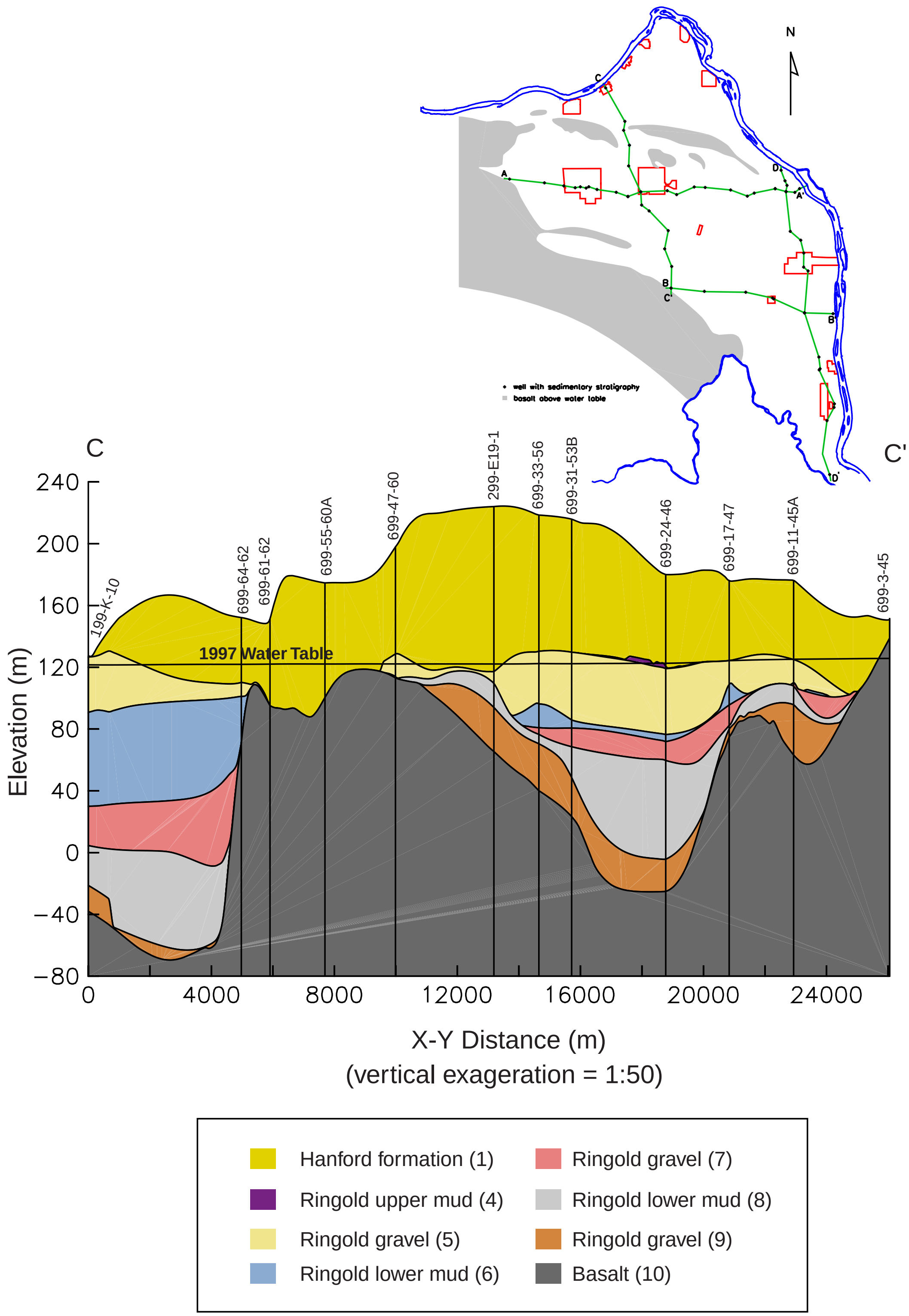
As the post-basalt sediments were being deposited, the Pasco Basin continued to undergo structural deformation (DOE 1988). The basin continued to subside, and the ridges continued to rise. This process caused sedimentary units to be thickest in the center of the basin and thin or, in places, pinch out along the anticlines, as illustrated in Figure D-8. Hanford formation sediments directly overlie the basalt in a few places where the Ringold Formation either was never deposited or was eroded by the ancestral Columbia River and its tributaries before the Missoula floods, as illustrated in cross-section C-C' (Figure D-8). Missoula floodwaters further eroded sediment and basalt in some areas. Two known vertical faults, the Cold Creek and May Junction faults (Figure D-6), were also developing as the older Ringold sediments were being deposited. Faulting is thought to have occurred until middle Ringold time, with a maximum vertical offset of 150 m; there is no evidence of activity on these faults since that time (Johnson et al. 1993). The Cold Creek fault is known to affect hydraulic heads in the confined basalt aquifers; however, it is not clear if the unconfined aquifer is also affected.

The unconfined aquifer, and a sequence of confined aquifers, lie beneath most of the Hanford Site. The unconfined aquifer is generally located in the unconsolidated to semiconsolidated Ringold and Hanford formation sediments that overlie the basalt bedrock. Where it is below the water table, the coarse-grained Hanford formation comprises the most permeable zones of the unconfined aquifer system. The basalt confined aquifers are composed of the brecciated tops of basalt flows and sedimentary interbeds located between basalt flows of the Columbia River Basalt Group.

The saturated thickness of the unconfined aquifer on the Hanford Site is greater than 61 m in some areas, but pinches out along the flanks of the basalt ridges. Depth to the water table ranges from less than 0.3 m near the Columbia River to more than 106 m near the 200 Areas. Perched water-table conditions have been encountered in sediments above the unconfined aquifer in the 200 West Area (Airhart 1990, Last and Rohay 1993) and in irrigated offsite areas east of the Columbia River (Brown 1979).

Groundwater, in the unconfined aquifer at the Hanford Site, generally flows from recharge areas in the elevated regions near the western boundary of the Hanford Site toward the Columbia River. The Columbia River is a discharge zone for the unconfined aquifer on both sides of the Columbia River. The Yakima River lies southwest of the Hanford Site, and is generally regarded as a source of recharge to the unconfined aquifer in the southern part of the Hanford Site and in the Richland area. Areal recharge from precipitation is highly variable both spatially and temporally, depending on climate, soil type, and vegetation.

Figure D-8. Geologic Cross-Sections Through the Hanford Site.



D.2.1.2 Flow System Boundaries. The unconfined aquifer system at the Hanford Site is bounded by the Columbia River on the north and east and by the Yakima River and basalt ridges on the south and west (Figure D-9). River stage and groundwater level observations associated with these two river systems provide good rationale for assuming they should be defined as prescribed head boundaries. The basalt ridges on the south and west are assumed to be no-flow boundaries. At the Cold Creek and Dry Creek Valleys, the unconfined aquifer extends westward beyond the boundary of the Hanford Site groundwater flow model. Boundaries have been defined across these valleys. An arbitrary boundary has also been defined between Umtanum Ridge and the Columbia River in the northwest corner of the Hanford Site. The upper boundary of the unconfined aquifer is usually the water table, which changes position over time. However, fine-grained sediments in a few areas may locally confine the aquifer. Flow through the upper boundary includes both natural areal recharge from precipitation and local recharge from liquid waste disposal, irrigation, and artificial recharge activities. Discharge through wells is minor and has not been included in the numerical model. The Richland city well field has a net recharge because of the input of Columbia River water to recharge basins. The bottom of the unconfined aquifer system is generally defined as the top of basalt. Any recharge or discharge through this boundary results from interflow with the underlying confined aquifer system.

D.2.1.2.1 Columbia River Boundary. The Columbia River flows along the northern and eastern boundaries of the modeled portion of the Hanford Site. Groundwater in both the unconfined and confined aquifer systems generally flows toward the river, which is the major discharge area within the Pasco Basin.

For the base conceptual groundwater model, the Columbia River boundary will be assumed to be best represented as a prescribed-head boundary over the depth of the river and as a no-flow boundary from the bottom of the river to the bottom of the aquifer, as shown in Figure D-10. Groundwater in the unconfined aquifer system is unlikely to flow across this boundary because portions of the river are believed to be the regional discharge for the Pasco Basin. However, flow across this boundary is possible if a locally confined permeable unit extends beneath the river and is affected by stresses (i.e., pumping). Definitions of hydrogeologic units in the conceptual model are being extended across the river to allow for possible simulations of such a scenario or other scenarios where local flow may pass under the river before discharging to it.

Water levels in many wells near the Columbia River fluctuate in response to changes in river stage. The river stage generally rises and falls daily because releases from upstream dams can change local river levels by up to 3 m within a few hours. Seasonal changes of about the same magnitude are also observed. River stage fluctuations measured at the 300 Area are only about half the magnitude of those measured near the 100 Areas because of the effect of the pool behind McNary Dam, located downstream from the Hanford Site (Campbell et al. 1993). Changes in water-table elevation near the river result primarily from pressure waves transmitted through the unconfined aquifer. However, some water also moves into the aquifer from the river during high river stage resulting in “bank storage” effects. Hydrographs showing the influence of the river stage on the unconfined aquifer (at various locations along the Columbia River) are presented by Newcomb and Brown (1961), Jensen (1987), Liikala et al. (1988), Schalla et al. (1988), Fruland and Lundgren (1989), Luttrell et al. (1992), McMahon and Peterson (1992), and Campbell (1994). For a general site-wide model, daily and seasonal changes in the river stage resulting

Figure D-9. Hydraulic Boundaries of the Unconfined Aquifer.

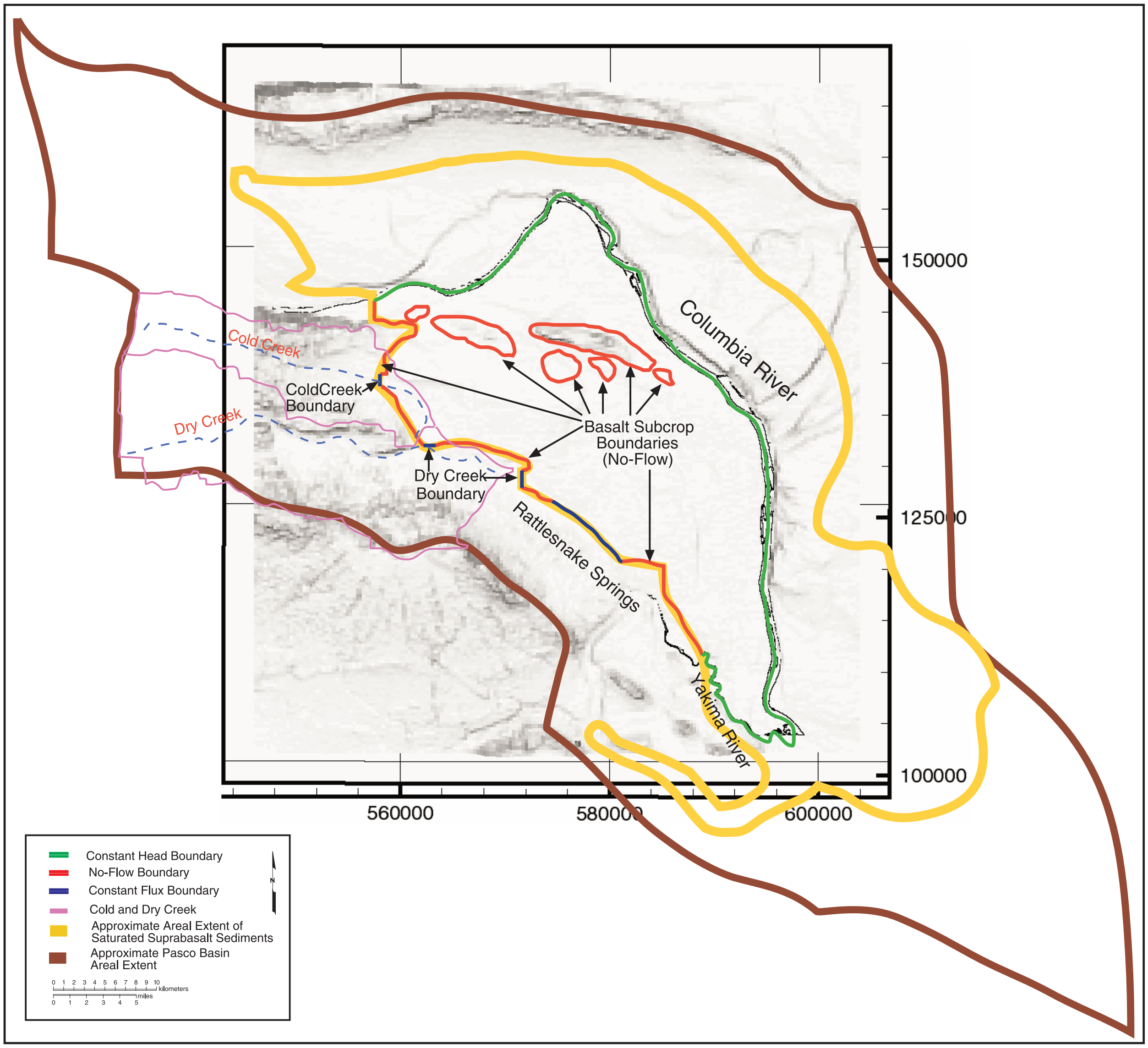
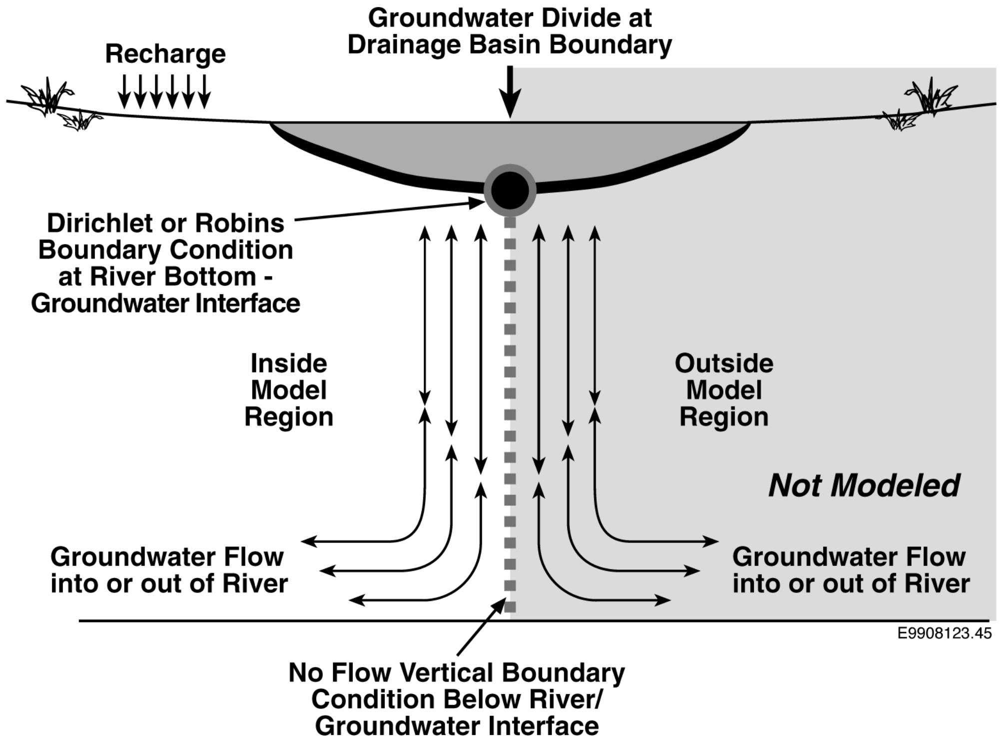


Figure D-10. Conceptualization of Boundary at the Columbia River.



from releases from upstream dams can be ignored, and a time-averaged river stage can be used for the prescribed-head value at the river.

Measurements of actual groundwater flux to the Columbia River would be extremely valuable for model calibration. However, because of the large flow of the Columbia River compared to the contribution from groundwater, measurements of the relatively small flow-rate changes expected to occur along the Hanford Reach are not feasible with any known technology. Estimates of groundwater discharge to the Columbia River have been made in past studies. Luttrell et al. (1992) applied a flow net analysis to calculate discharge near the old Hanford Townsite. They estimated 6.6 x 10^6 m^3/yr discharge to about a 1-km length of the river. An earlier estimate of 2.7 x 10^6 m^3/yr for the same area was based on the site-wide flow model (Prater et al. 1984). In comparison, the average annual Columbia River flow is about 1.06 x 1011 m^3/yr.

D.2.1.2.2 Yakima River Boundary. The Yakima River borders the southeastern corner of the modeled area for a distance of approximately 25 km. This area includes the western edge of the southern end of the Hanford Site and the western edge of the city of Richland. The Yakima River has usually been represented by a prescribed-head boundary in previous models (Jacobson and Freshley 1990). Because the water levels in the river are higher than the heads within the adjacent aquifer, the river is a potential source of recharge. The recharge rate is controlled by the hydraulic conductivity of sediments adjacent to the river and the head difference between the river and aquifer. However, the recharge at this boundary is highly uncertain because of a lack of wells and a corresponding lack of information concerning hydraulic properties and water-level elevations near the river. This causes uncertainty in the model predictions of groundwater flow within the area between the Yakima and Columbia Rivers, an area that is becoming increasingly important as commercial development continues immediately south of the Hanford Site boundary.

As part of a study of groundwater chemistry of the Pasco Basin (Ebbert et al. 1993), the U.S. Geological Survey found evidence that the Yakima River recharges the unconfined aquifer in the Hanford Reach adjacent to the Hanford Site. This conclusion was based on a comparison of the chemical composition of river water, groundwater from a well completed in the Saddle Mountains Basalt, and groundwater from an offsite well completed in the unconfined aquifer (Ringold Formation) near the river.

To help define aquifer behavior near the Yakima River, river-stage monitoring has been conducted just below Horn Rapids Dam. As reported in Thorne et al. (1993), water levels were continuously monitored at well 699-S24-19 for both the unconfined aquifer system and the basalt-confined aquifer system. The water levels at this well did not show a direct response to changes in river stage. However, the water level of the unconfined aquifer interval does respond to the filling of a canal (the Horn Rapids Ditch) between the well and the river.

The section of the Yakima River below Horn Rapids Dam flows through flood-plain sediments that consist of moderately permeable stream channel deposits within fine-grained overbank and oxbow lake deposits. In this area, the unconfined aquifer may be somewhat isolated from the river by these fine-grained deposits. Examination of drilling logs for private wells near the river shows that there is often fine-grained material near the water table, which sometimes acts as a locally confining unit. After water-bearing sediments are encountered, the water level in the well rises into the depth interval corresponding to the fine-grained material. The presence of low-permeability sediments near the river would also explain the lack of water-level response to the river stage at well 699-S24-19. However, because this well responds to filling of the canal, which is closer to the well, it is likely that the low-permeability sediments do not extend to the canal location. Recent silt deposits in the bed of the river could also explain the lack of response.

D.2.1.2.3 Cold Creek Valley. The boundary of the model region crosses the Cold Creek Valley at the northwestern corner of the Hanford Site. This is an area where the model boundary does not coincide with a physical boundary of the unconfined aquifer flow system. The unconfined aquifer sediments extend into the valley and are a conduit for recharge to the Hanford Site aquifer system. Actual recharge quantities from Cold Creek Valley are not known. Jacobson and Freshley (1990) used a prescribed-flux boundary with an assumed recharge of about 9,100 m^3/d at the mouth of the Cold Creek Valley in two of the cases they ran for the inverse calibration model. The result in both cases was unrealistically high head values calculated by the model near Cold Creek Valley. Therefore, either the prescribed recharge at this boundary was too large or transmissivities in the area were set too low. Jacobson and Freshley (1990) obtained better results when using a prescribed-head boundary. However, uncertainty in the transmissivity distribution remains because it is not known if the recharge calculated by the model at this boundary, which depends on the hydraulic gradient across the boundary and the transmissivity of the adjacent model elements, is realistic.

A hydraulic test was conducted at well 699-43-104 during 1994. This test resulted in a relatively low transmissivity estimate of 25 m^2/d and an equivalent hydraulic conductivity of about 2 m/d. However, these values may not be representative of the bulk of Cold Creek Valley sediments.

D.2.1.2.4 Interflow with the Basalt Confined Aquifer System. Flow-system boundaries are formed by the contact between the unconfined aquifer system and basalt. At places where basalt subcrops above the water table, this contact may form either a perimeter boundary or an island of basalt within the model area. The basalt contact also forms the lower boundary of the unconfined aquifer system, except in some areas where a mud unit may underlie the aquifer directly over basalt.

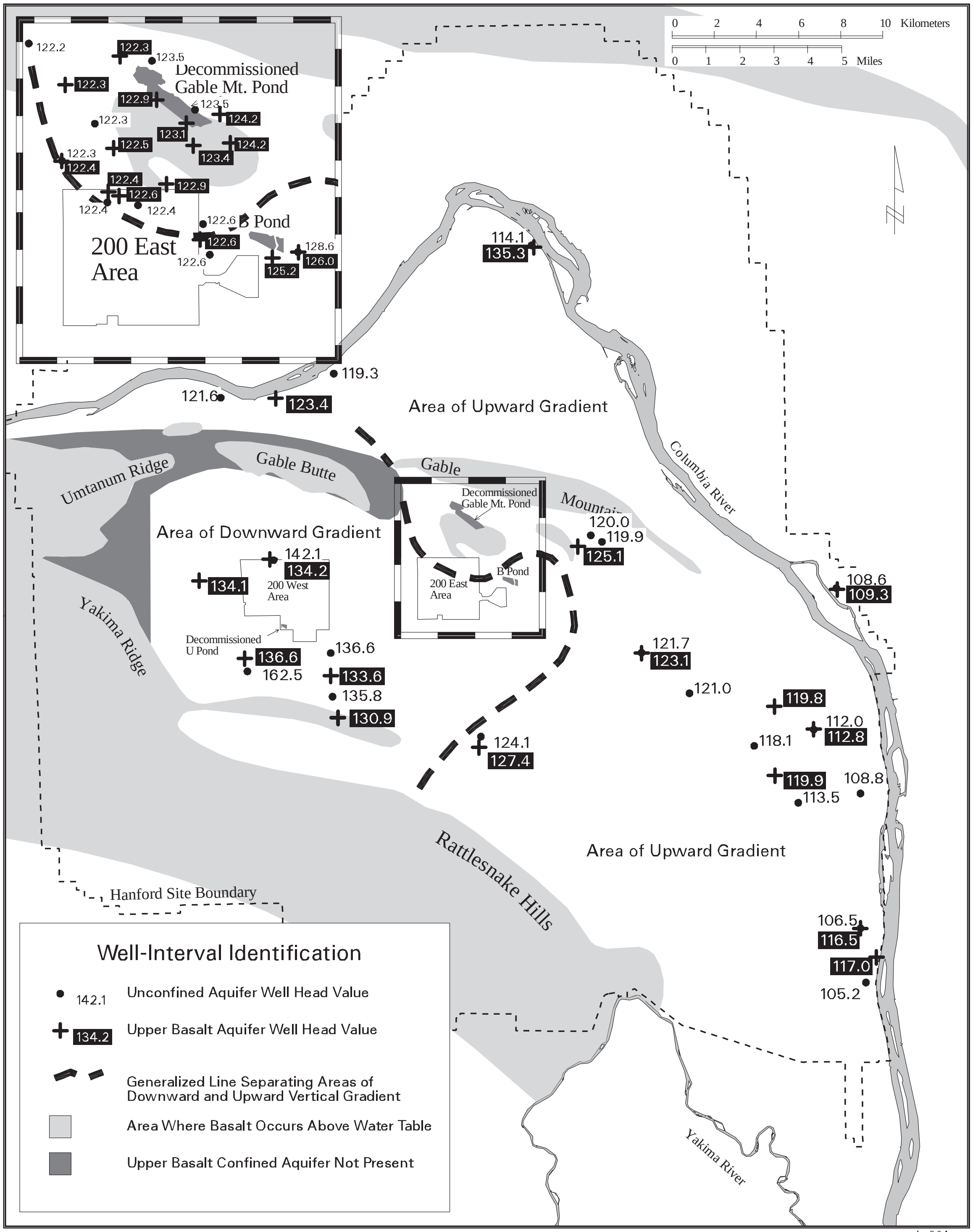
Some of the perimeter basalt contact boundaries (i.e., Rattlesnake Mountain) may be recharge boundaries because of the infiltration of precipitation runoff and spring discharge from the upper slopes. There is also a potential for interflow (recharge or discharge) between the basalt confined aquifer system and the unconfined aquifer system at the lower boundary. Throughout most of the Hanford Site, the amount of interflow is thought to be small because of the low hydraulic conductivity of the rock separating the two aquifer systems. However, areas of increased vertical flow have been previously identified in the Gable Mountain and Gable Butte area on the basis of chemistry data (Graham et al. 1984, Jensen 1987). Hydraulic head data for the uppermost confined basalt aquifer also indicates the potential for water to discharge from this aquifer upward into the unconfined system in the northeastern part of the Hanford Site (Spane and Raymond 1993; Spane and Webber 1995). Figure D-11 shows a comparison of observed hydraulic heads for the two aquifer systems based on water-level data collected in 1994 and delineates areas of upward and downward hydraulic gradient.

Another potential area of increased vertical flow between aquifers is near the Yakima River horn, where the river has incised the upper basalt confining layers. A recent investigation (WHC 1993) identified a bimodal distribution of chloride in the unconfined aquifer in this area. Some wells yield concentrations of less than 10 mg/L, and other wells have greater than 20 mg/L. The lower concentration groundwater is chemically similar to water from Rattlesnake Hills springs, suggesting that this groundwater comes from subsurface discharge from the underlying basalts. The groundwater with higher chloride concentrations may come from infiltration of surface flow, which is subject to greater evaporation.

Interflow between the unconfined and basalt confined aquifer systems is assumed to be insignificant in the proposed base conceptual model. The rate of groundwater movement between the confined and unconfined aquifer systems is difficult to quantify. Therefore, it is not known if ignoring this contribution has a significant effect on the accuracy of the groundwater flow model. Differences in groundwater chemistry and temperature offer two possible methods for identifying areas of enhanced interflow and possibly quantifying flow rates. The possible use of temperature logs has been preliminarily investigated, and results are presented in Thorne et al. (1994).

D.2.1.3 Recharge and Discharge. Natural recharge to the unconfined aquifer system occurs from infiltration of runoff from 1) elevated regions along the western boundary of the Hanford Site; 2) infiltration of spring water that originates from the basalt aquifer system; and

Figure D-11. Comparison of Observed Heads for the Upper Basalt and Overlying Unconfined Aquifer.



3) infiltration of precipitation falling across the Hanford Site. Some recharge also takes place along the Yakima River, in the southern end of the Site. Since the start of Hanford Site operations in the mid-1940s, the estimated recharge from these natural sources has been less than the artificial recharge from waste-water disposal facilities. However, during the past five years, most production activities on the Hanford Site have been curtailed resulting in a decrease in waste-water disposal. Currently, the volume of artificial recharge is similar to the volume of natural estimated recharge (Fayer and Walters 1995).

The Columbia River is the principal discharge area for the unconfined aquifer system. A few wells produce water from the unconfined aquifer on the Hanford Site. However, the total volume produced is relatively small and is not expected to be a significant discharge component on the site-wide scale. The supply wells serving the 400 Area have the highest withdrawal rates, which average about 500 m^3/d.

D.2.1.3.1 Natural Areal Recharge. Natural areal recharge from precipitation is highly variable both spatially and temporally, ranging from near zero to more than 100 mm/yr depending on climate, vegetation, and soil texture (Gee et al. 1992, Fayer and Walters 1995). Areas with shrubs and fine-textured soils like silt loams tend to have low recharge rates, while areas with little vegetation and coarse-textured soils (i.e., dune sands) tend to have high recharge rates. Recharge is also generally higher near the basalt ridges because of greater precipitation and runoff. Past estimates of recharge have been summarized in earlier status reports (Thorne and Chamness 1992, Thorne et al. 1993). Natural recharge estimates for 1992 were developed by Fayer and Walters (1995). Fayer and Walters (1995) first mapped the distributions of soil and vegetation types. A recharge rate was then assigned to each combination on the basis of data from lysimeters, tracer studies, neutron probe measurements, and computer modeling. Estimated recharge rates for 1992 ranged from 2.6 to 127 mm/yr, and the total volume of natural recharge from precipitation over the Hanford Site was estimated at 8.47×10^6 m^3/yr. This value is of the same order of magnitude as the artificial recharge to 200 Area waste disposal facilities during 1992 and is about half the volume of discharge to these facilities during 1979 (Fayer and Walters 1995). Similar estimates of recharge were developed for 1979 conditions and are illustrated in Figure D-12. The 1979 recharge estimate assumes that any effects of Hanford Site operations, which may influence recharge, had not reached the water table by 1979. The 1979 recharge distribution is significantly different from the 1992 estimate in part because of this consideration, but also because there were major changes in the vegetation distribution due to a fire at the Hanford Site in 1984. The 1979 recharge estimate is the assumed present and future condition for the draft base case groundwater conceptual model.

D.2.1.3.2 Artificial Recharge. The large volume of wastewater discharged to disposal facilities on the Hanford Site during the past 50 years has significantly affected the groundwater flow system. As shown in Figure D-13, the volume of artificial recharge has decreased significantly during the past 10 years and is currently still decreasing (Barnett et al. 1995, Dresel et al. 1995). Until it was removed from service in 1984, Gable Mountain Pond received the largest volume of discharge on the Hanford Site. Major groundwater mounds have occurred beneath B Pond, Gable Mountain Pond, and U Pond, and have affected site-wide groundwater flow patterns (Bierschenk 1959, Dresel et al. 1995). Wastewater is no longer being discharged to U Pond and

Figure D-12. Estimated Annual Recharge from Infiltration of Precipitation Based on 1979 Conditions.

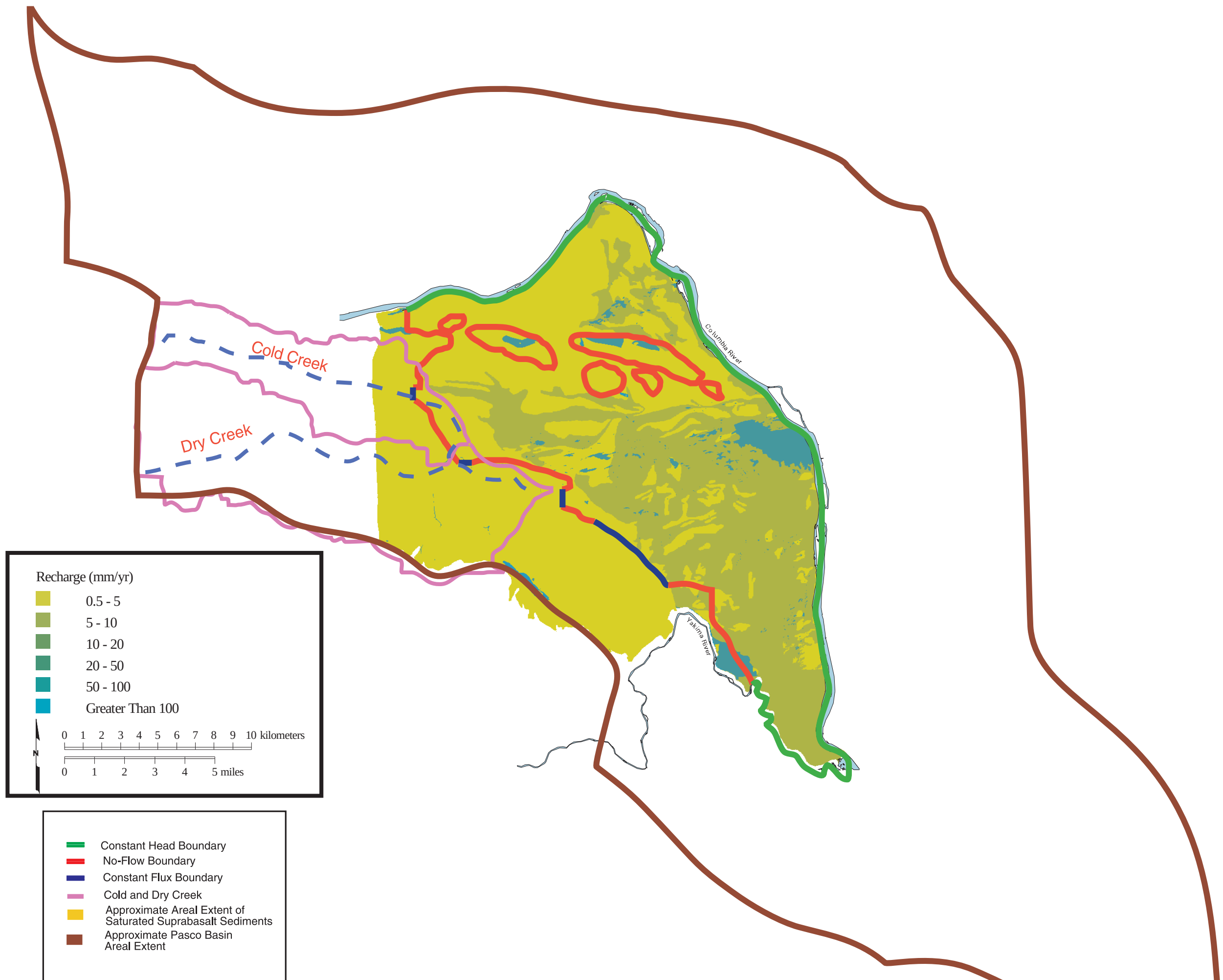
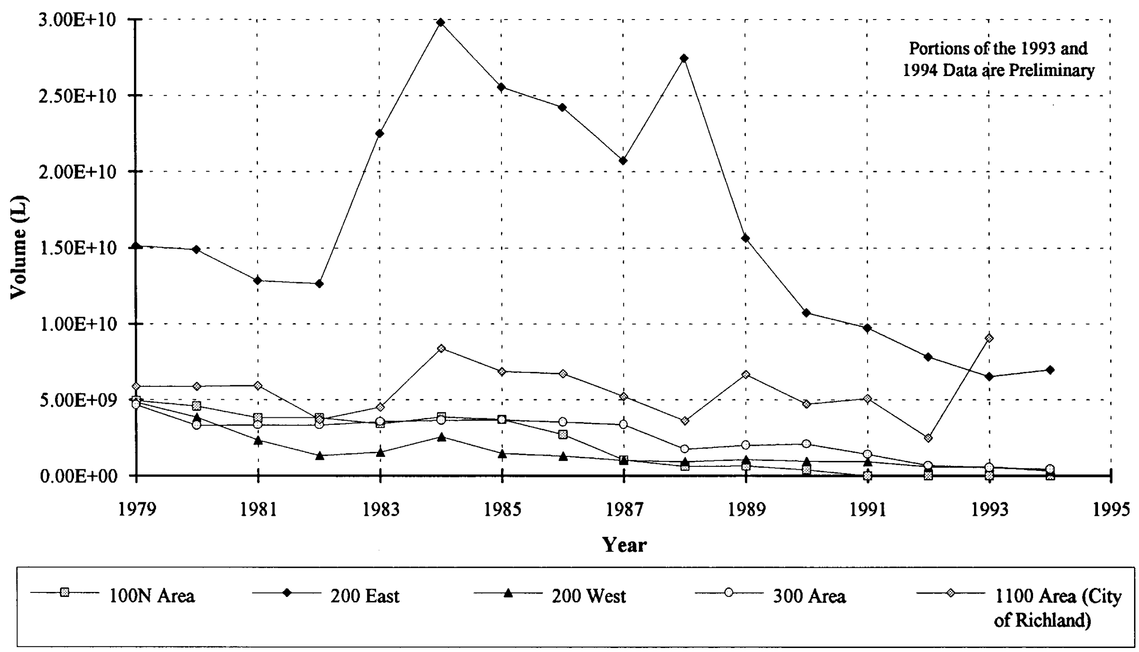


Figure D-13. Hydrograph of Waste Streams at the Hanford Site 1979 to 1994.



Gable Mountain Pond, which have been decommissioned and are now dry. Other smaller volume recharge sources have existed until recently in the 100, 200, and 300 Areas and may affect groundwater flow on a local scale. The B Pond was decommissioned in 1997. Currently, all tritiated water is disposed to the State-Approved Land Disposal Site (SALDS), while the major artificial recharge source of clean water is the 200 Area Treated Effluent Disposal Facility (TEDF). Additional information on wastewater discharge is available in the *Hanford Site Groundwater Protection Management Plan* (Barnett et al. 1995).

The city of Richland infiltration ponds, agricultural and lawn irrigation, and ground disposal of wastewater at a potato-processing plant are other sources of artificial recharge that may affect groundwater flow in the north Richland area and in the southern part of the Hanford Site (Liikala 1994).

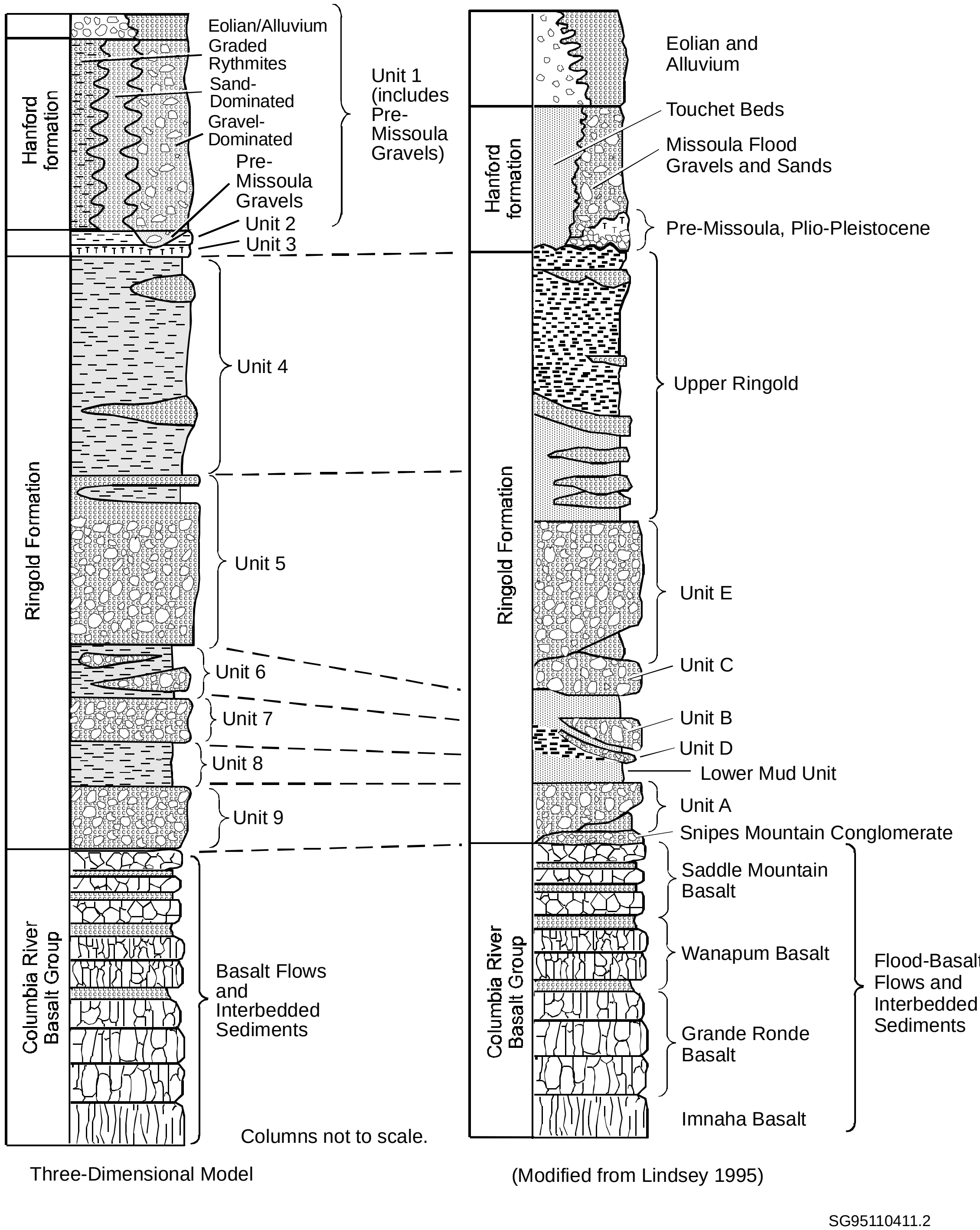
D.2.1.4 Hydrogeologic Unit Identification and Characteristics. Understanding the lateral extent and relationships between the hydrogeologic units found in different parts of the Hanford Site is crucial to understanding the movement of groundwater contaminants and for constructing accurate contaminant transport models. For example, it is important to determine whether or not fine-grained units found in the eastern and western portions of the Hanford Site directly overlap one another in the central part of the basin to form a continuous aquitard. The units proposed for the base conceptual model are discussed in the following sections. In total, nine units numbered one through nine are proposed. The five odd-numbered units consist of predominately coarse-grained sediments. The four even-numbered units consist of predominately fine-grained sediments.

D.2.1.4.1 Identification of Hydrogeologic Units. The movement of groundwater within the aquifer is controlled by hydraulic conductivity, which is closely related to the sediment texture. Texture is a function of the grain-size distribution, sorting, and consolidation/cementation. Sediments were differentiated into either coarse or fine texture groups, then split into individual hydrogeologic units based on stratigraphic position, color, and distinctive markers such as ash horizons. Normally, identifying geologic units also uses depositional environment and relative time of deposition to define contacts between units. Because the prime interest is in the movement of groundwater, the important geologic information is related to the movement of groundwater. Figure D-14 shows a comparison of a geologic stratigraphic column and the proposed units for the base conceptual model. The two are very similar, but it is important to clarify the difference. An example is the lower part of the upper Ringold, as defined by Lindsey (1992), which in some places becomes progressively more sandy with depth. Where sand is the only (or overwhelmingly dominant) grain size, it was grouped with the underlying coarse-grained Unit 5. Although this may not conform to standard geologic classification, the sandy base of the upper Ringold is probably hydraulically connected with and hydrologically similar to Unit 5, with which it is proposed to be grouped in the base conceptual model. Generally, sands were grouped with sandy gravels, and silt was grouped with clay, assuming similar hydraulic conductivities.

Similar units were identified in studies focused on operational areas. Individual reports on the 100 Areas (Peterson 1992; Hartman and Lindsey 1993; Lindberg 1993a, 1993b; Lindsey and Jaeger 1993), the 200 Areas (Connelly et al. 1992a, 1992b), and the 300 Area (Swanson 1992) have been released in the past five years, but few geologic studies have addressed the regions of the Hanford Site lying between the operational areas. The nine hydrogeologic units identified for the proposed base conceptual model are similar to those in the previous reports, with some differences in the location of unit contacts in places, as previously discussed and shown in Figure D-14. The column on the left side of Figure D-14 shows the hydrogeologic layering proposed for the base case groundwater conceptual model, where the individual layers are established based on similarities in their hydraulic characteristics rather than their geology. The column on the right is the standard Hanford Site stratigraphic column and the dashed lines connecting the two columns illustrate how the nine hydrogeologic layers map to the standard Hanford Site stratigraphy.

D.2.1.4.2 Determination of Geologic Contacts. Data from 426 wells across the Hanford Site have been used to define hydrogeologic units based on textural composition, as described in Wurstner et al. (1995). Top of basalt was identified in an additional 150 wells. Data used to define hydrogeologic units included well logs, downhole geophysical logs, particle size analyses, calcium carbonate content, and geologic interpretations from other reports. Once the distribution of each hydrogeologic unit was understood, a line showing the estimated extent of each unit was generated (i.e., lines showing where each unit reaches zero thickness). These "extent lines" were made as accurate as possible on the west and south sides of the Hanford Site, where the units pinch out on the basalt highs near the edge of the model. To the north and east, however, the

Figure D-14. Comparison of Generalized Geology and Hydrogeologic Stratigraphic Columns.



conceptual model extends only as far as the Columbia River and the basalt highs are much further away. Consequently, in those areas the extent lines were drawn to some arbitrary distance beyond the river to create an appropriate thickness for each unit at the edge of the model beneath the river. The gridding program interpolated the data set beyond the actual lateral extent of the unit, and the model boundary was used to truncate the interpolated 2-D grids, where necessary. A brief description of each unit follows from Wurstner et al. (1995). The lateral extent and thickness of each unit is based on data and inferences from wells, as described previously. The distribution of these wells for each unit and isopach maps, as interpreted from the data are provided in Wurstner et al. (1995).

The uppermost basalt flow was used as the bottom of the hydrogeologic framework because it forms the base of the Hanford Site unconfined aquifer. Unit 9 lies directly above basalt at the bottom of the unconfined aquifer system. This unit consists of fluvial sand and gravel and generally correlates to Unit A (Lindsey 1992) (basal Ringold). Unit 9 is found in the deeper parts of the basin, pinching out on (or eroded from) the limbs of the basalt anticlines. In most places, Unit 9 is overlain by Unit 8. Unit 8 is equivalent to the Lower Mud Sequence (Lindsey 1992) (the lower Ringold and part of the basal Ringold) and forms an aquitard across much of the Hanford Site. The mud in this unit is often described as blue or green, sticky clay, and frequently includes a white “ash” that may correspond to the ash in the lower Ringold in Bjornstad (1984). Unit 8 is relatively extensive across the Hanford Site.

Units 6 and 7 have more complex relationships and are more difficult to classify. When these units were deposited, the river channel apparently shifted position more often, depositing a complex pattern of overbank and mainstream deposits. To simplify the conceptual model, Unit 7 is defined as the coarse-grained sediments immediately overlying Unit 8. Unit 6 is defined as the sequence of mostly fine-grained sediments with some interbedded coarse-grained sediments overlying Unit 7 and underlying Unit 5. Unit 7 generally corresponds to Units B and D (Lindsey 1992), while Unit 6 corresponds to Unit C and the unnamed mud layers.

Where coarse-grained Unit 7 is not present, Units 6 and 8 cannot usually be distinguished. In these cases, the fine-grained sediments are usually grouped into Unit 8. Likewise, where fine-grained Unit 6 is not present, Units 5 and 7 cannot be distinguished, and the coarse-grained sediments are grouped into Unit 5. Unit 5 corresponds to the fluvial, coarse-grained sediments of the Unit E (Lindsey 1992) (middle Ringold). This unit is quite thick in the western portion of the Hanford Site where Units 6 and 7 are not recognized. In many parts of the Hanford Site, the water table is presently found in Unit 5.

Overlying Unit 5 is Unit 4, a fine-grained fluvial and lacustrine unit that corresponds to the Upper Ringold Unit (Lindsey 1992). Unit 4 has been eroded from large portions of the Hanford Site. In the eastern part of the area north of Gable Mountain, distinction between the fine-grained Unit 6 and the probable base of Unit 4 cannot be made, and the sediments are all grouped into Unit 6.

Units 2 and 3 correspond to the early “Palouse” soil and the Plio-Pleistocene unit, respectively. Unit 3 is a buried soil horizon containing caliche and side-stream basaltic gravels. It is only recognized in the western part of the basin. The caliche developed on the top of the eroded

Ringold sediments and has a low hydraulic conductivity, while the side-stream gravels have a high conductivity. There is only one small area south of the 200 West Area where Unit 3, as the side-stream gravels, intersects the water table. Unit 2 is a small pocket of fine-grained sediments that have been interpreted as eolian silt.

Unit 1 is the Hanford formation, which is generally a high permeability sand and gravel unit that covers most of the Hanford Site. In most areas where Unit 1 is below the water table, the sediments are gravels or coarse sands. The finer grained sand- and silt-dominated facies are mostly above the water table within the boundaries of the Hanford Site. The surficial sand dunes have been included with Unit 1.

Lying beneath the gravels of the Hanford formation, in the central portion of the Hanford Site, are the sand and gravel deposits commonly called the "pre-Missoula gravels" (PSPL 1982). These sediments have been grouped with the Hanford formation (Unit 1) for the following reasons: 1) the pre-Missoula gravels cannot be readily distinguished from the Hanford formation in most driller's or geologist's logs; 2) there are no known hydraulic property data for the pre-Missoula gravels, although its properties probably lie between the younger Hanford gravel-dominated facies and older sandy gravel of Unit 5; and 3) the pre-Missoula gravels are above the water table except in some areas near the Hanford Townsite and near the solid waste landfill in the center of the Hanford Site. Therefore, they do not present a primary pathway for groundwater movement.

D.2.1.5 Hydraulic Parameters. Hydraulic properties, including both horizontal and vertical hydraulic conductivity (K_h and K_v), storativity (S), and specific yield (S_y), are key components of the conceptual groundwater model. The distribution of these parameters must be specified for each hydrogeologic unit. Hydraulic conductivity controls the rate of water flow through a unit thickness of the aquifer at a given hydraulic gradient. Storativity and specific yield determine the change in water-table elevation that will occur in response to a change in the volume of water stored in the aquifer.

Hydraulic property data for the Hanford Site unconfined aquifer have been derived mainly from aquifer pumping tests and, in a few cases, from laboratory permeameter tests. These results have been documented in dozens of published and unpublished reports during the past 50 years.
A summary of available data for the unconfined aquifer was provided in DOE (1988), and an updated summary was provided in Thorne and Newcomer (1992), together with an evaluation of selected pumping test analyses. Additional tests have been conducted both to support the 3-D model and to support other Hanford Site projects. Some of the recent tests are documented in status reports on the development of the 3-D conceptual model (Thorne and Chamness 1992; Thorne et al. 1993, 1994).

During 1995, a pumping test was conducted by the city of Richland on a new water supply well located near Wellsian Way in the southern part of Richland. Data were collected from a nearby observation well and analyzed to provide an additional measurement of hydraulic properties for the Hanford formation (Unit 1) in this area. The test analysis and results are described in Wurstner et al. (1995).

Newcomb and Strand (1953) analyzed the growth of groundwater mounds beneath liquid disposal facilities in both the 200 West Area and 200 East Area between 1948 and 1953 to estimate hydraulic properties for these areas. Recent decreases in disposal volumes have caused a decrease in these mounds that has been analyzed to obtain additional hydraulic property information. Details of the analysis of the mound dissipation are provided in Wurstner et al. (1995).

D.2.1.5.1 Hydraulic Conductivity of Hydrogeologic Units. Hydraulic conductivity values for sediments composing the unconfined aquifer system range from less than 10^{-4} m/d for some mud units to about 10^{6} m/d for coarse gravel flood deposits. The sand and gravel facies of the Ringold Formation are about 10 to 100 times less permeable than the coarse sediments of the overlying Hanford formation (DOE 1988). The Ringold Formation also contains relatively extensive layers of fine-grained, low permeability sediments (i.e., silt or clay).

Most pumping test analyses result in estimates of aquifer transmissivity (T), which, for a vertically homogeneous aquifer, is the product of hydraulic conductivity in the horizontal plane, K^h and aquifer thickness. A listing of available transmissivity data, obtained from pumping tests in the unconfined aquifer system, is provided in Wurstner et al. (1995); and Figure D-15 shows the distribution of the tested wells across the Hanford Site with the associated main geologic unit tested. The data provided in Wurstner et al. (1995) include 36 single well pumping tests and 3 multiple well pumping tests that pertain to the Hanford formation (Unit 1). Thirty-seven single well pumping tests and 12 multiple well pumping tests pertain to Ringold Formation sand and gravel units (Units 5, 7, and 9). An additional 32 single well pumping tests, 7 multiple well pumping tests, and 2 specific capacity tests, for which the tested hydrogeologic unit has not been defined, are included. The quality of these results is affected by both aquifer conditions and analysis procedures and varies widely (Thorne and Newcomer 1992). Slug tests have also been conducted at several Hanford Site wells. However, because many of the single well slug test results are considered inaccurate, they have not been used to determine hydraulic properties for the proposed base conceptual model. Multiple well slug tests have been conducted at a few wells in conjunction with multiple well pumping tests. Because of vertical aquifer heterogeneity, and because most of the tested wells at the Hanford Site partially penetrate the unconfined aquifer, it is sometimes difficult to determine the aquifer thickness that should be used in calculating hydraulic conductivity from the test results.

In both the permeable units and the less permeable units (i.e., even numbered units), single values of K_h are assigned, based on the assumption that hydraulic conductivity is isotropic in the horizontal plane. The uppermost permeable unit for most of the model region is either Unit 1 or Unit 5. Units 7 and 9 represent deeper permeable units. The hydraulic conductivity of Unit 1 generally ranges from about 1 to 1,000,000 m/d and is much higher than any of the other units that compose the unconfined aquifer system. Therefore, where it is present below the water table, this unit usually provides the dominant flow path within the aquifer. Figure D-16 outlines which geologic units contain the water table at the Hanford Site during 1998. Unit 1 consists of sand and gravel of the Hanford formation and the pre-Missoula gravel deposits. Extensive fine-grained facies of the Hanford formation are not found below the water table within the model region. Near B Pond, the saturated portion of the Hanford formation is composed of muddy

Figure D-15. Distribution of Wells by Unconfined Aquifer Unit where Hydraulic Property Measurements were Obtained.

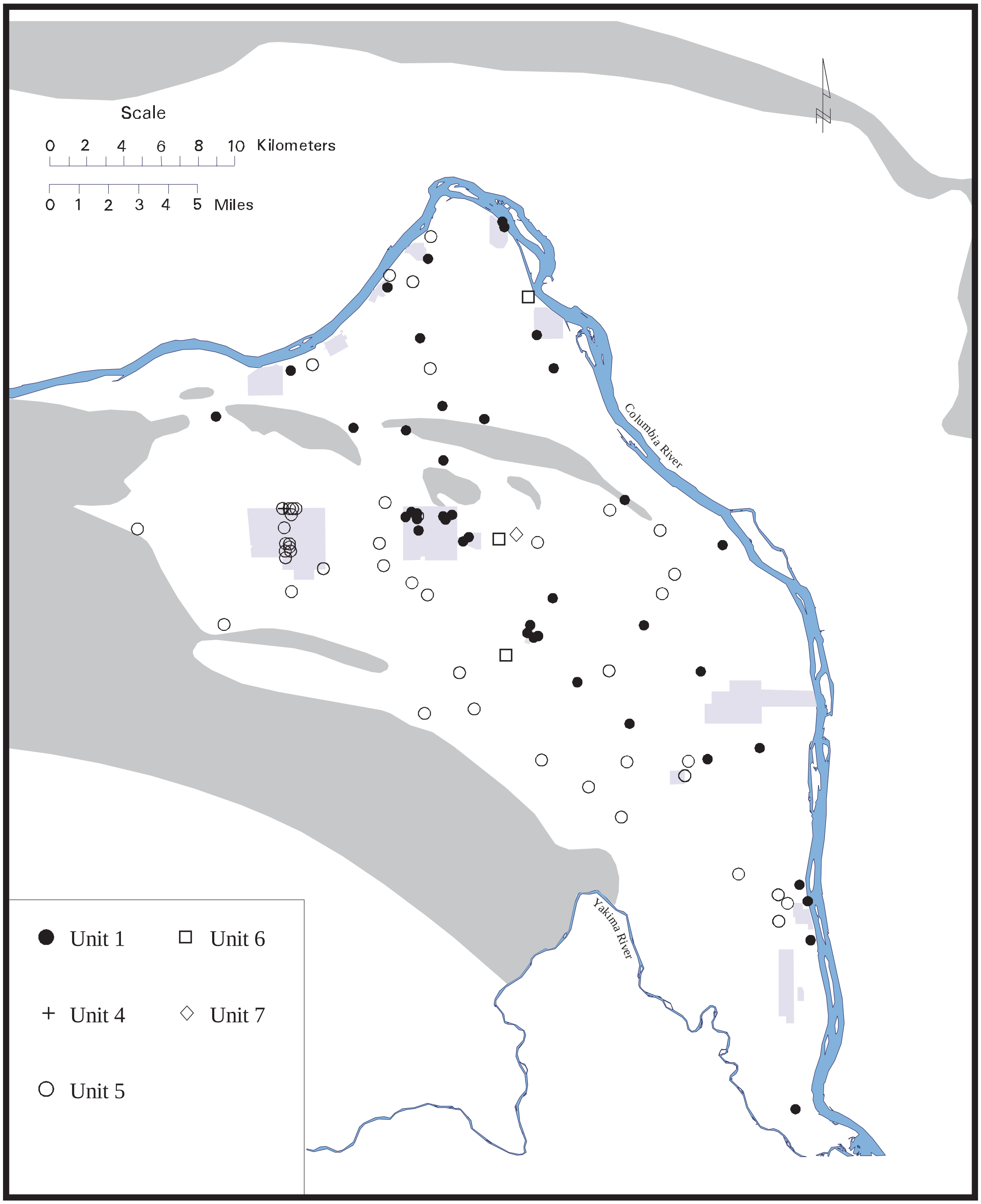
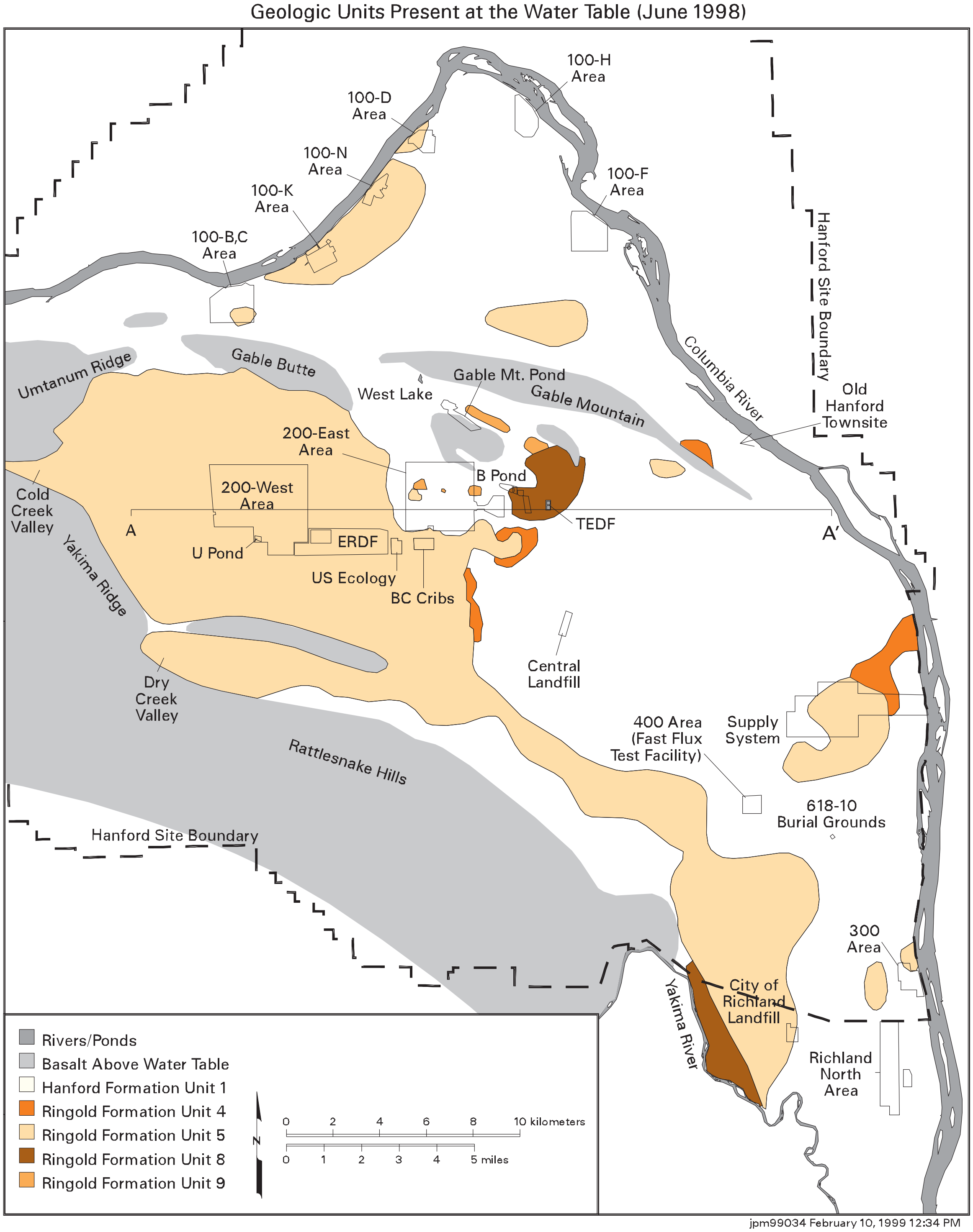


Figure D-16. Geologic Units Present at the Water Table, June 1998.



sandy gravels that probably represent the lower limit of hydraulic conductivity for Unit 1. Additionally, the water table is beginning to drop out of the Hanford formation in to the Ringold mud units east of the 200 East Area. Aquifer tests (Thorne et al. 1993) indicate that the minimum K_h is about 1 m/d, and the minimum K_v is about 0.02 m/d for Unit 1. The maximum measured value of K_h for Unit 1 on the Hanford Site is about 10,000 m/d (Thorne and Newcomer 1992, DOE 1988). However, the maximum hydraulic conductivity that can be measured by an aquifer test is limited by the well efficiency and the flow rate that can be pumped with available equipment. As a result, the upper limit of K_h for coarse gravel flood deposits of Unit 1 is probably greater than the values interpreted from existing field tests. Maximum K_v is unknown, but may approach the value for K_h in relatively clean gravel zones where stratified layers of finer grained material are not present.

Units 5, 7, and 9 are all within the Ringold Formation and consist of sand to muddy sandy gravel with varying degrees of consolidation and/or cementation. Unit 5 is the most widespread unit within the unconfined aquifer and is found below the water table across most of the model region. Hydraulic conductivities of Units 5, 7, and 9 determined from aquifer tests vary within the range of about 0.1 to 200 m/d. Because these units are hydrologically similar, they were grouped together in areas where the intervening mud units do not exist. A few aquifer tests suggest vertical anisotropy is in the range of 0.01 to 0.1. Therefore, the range of K_v is estimated at about 0.001 to 20 m/d.

Mud-dominated units within the unconfined aquifer system include Unit 4, also known as the upper Ringold fines; Unit 6, which is a composite of intercalated mud and sand and gravel layers; and Unit 8, which is an extensive lower Ringold mud unit. Hydraulic conductivity of these units is generally about 2 to 5 orders of magnitude less than that of the permeable sand and gravel units. Therefore, the mud units are essentially aquitards and are not expected to transmit significant quantities of water or contaminants in the horizontal direction. Mud units are most significant in slowing the vertical migration of contaminants and influencing vertical head distributions. Therefore, the values of K_v assigned to mud units are probably more important than the assigned values of K_h.

Hydraulic test results for mud-dominated units are listed in Table D-1. These few tests yielded hydraulic conductivity (K) values of 0.0003 to 0.09 m/d. Some of the results are from well tests and some are from laboratory tests. Because of a tendency to complete wells only in zones that are likely to produce some water, these values may represent the higher range of K_h for the mud units. Test results for Unit 6 indicate that this unit has higher K_h than Unit 4. This is expected because of the sand and gravel layers included in Unit 6. Unit 8 is expected to have hydraulic conductivity similar to Unit 4. Freeze and Cherry (1979) give a hydraulic conductivity range of 0.001 to 1 m/d for silt and loess, and as low as 10^{-7} m/d for clay. This range is partially based on a compilation of data by Davis (1969).

A summary of the current estimates of hydraulic parameters and ranges interpreted from field and laboratory test for the nine layers of the base case groundwater conceptual model is provided in Table D-2.

Table D-1. Hydraulic Test Results for Mud-Dominated Units.

Hanford Well Number	Hydraulic Conductivity (K) (m/d)	Hydrogeologic Unit
299-W7-9	0.09	Unit 4 (vadose)
699-20-39	<0.06	Unit 6
699-84-35A	0.03	Unit 6
699-41-40	0.0003	Unit 4

Table D-2. Summary of Current Estimates of Hydraulic Parameter Interpreted from Field and Laboratory Tests for the Nine Layers of the Base Case Groundwater Conceptual Model.

Layer Number[1]	Horizontal Hydraulic Conductivity[2], K_h (m/d)	Storativity (dimensionless)	Specific Yield (dimensionless)	Comments
1	10 to >3.5E+3	0.001 to 0.005	0.2 to 0.37	The water table surface is present in this layer in most of the eastern portion of the Hanford Site as shown in Figure D-16.
2	N/A	N/A	N/A	Currently, where this unit occurs on the Hanford Site, the water table is below this unit.
3	N/A	N/A	N/A	Currently, where this unit occurs on the Hanford Site, the water table is below this unit.
4	0.0003 to 0.09	no data	no data	The hydraulic conductivity is assumed to be a constant value in this layer.
5	0.1 to 560	0.0001 to 0.06	0.05 to 0.37	This layer is occurs at the water table in the western portion of the Hanford Site, as shown in Figure D-16.
6	0.002 to 0.03	no data	no data	Layer 6 is not present in western portions of the Hanford Site.
7	no data except composite zones, assume similar to Unit 5	no data	no data	The hydraulic conductivity is assumed to be a constant value in this layer.
8	no data	no data	no data	The hydraulic conductivity is assumed to be a constant value in this layer.
9	8 (only one test)	0.002	0.15	

[1]Layer numbers correspond to those shown on Figure D-10.
[2]The ratio K_h/K_v is assumed constant within each layer.
N/A = not applicable.

D.2.1.5.2 Storativity and Specific Yield. Storativity and specific yield can be calculated from multiple well pumping tests and multiple well slug interference tests (Spane 1993, 1994). The average specific yield from these tests was 0.15. However, some of these estimates are highly uncertain because of the effects of nonideal test conditions, such as partially penetrating wells and aquifer heterogeneity. Such conditions generally have a more significant effect on determining storage properties than determining of transmissivity. Moench (1994) demonstrated that these conditions can affect specific yield values calculated from type-curve analysis of aquifer pumping tests, and usually result in the calculated values being low.

Specific yield can also be calculated by measuring the change in saturated aquifer volume in response to the injection or withdrawal of a known volume of groundwater. This method was applied to the decreasing groundwater mound that occurred beneath the 200 West Area between 1985 and 1995 (Wurstner et al. 1995). The calculated specific yield was 0.17, which is higher than the 0.11 value calculated by Newcomb and Strand (1953), when they analyzed the growth of groundwater mounds beneath liquid disposal facilities in both the 200 West Area and 200 East Area between 1948 and 1953. The accuracy of results from both these analyses is uncertain, because the analyses assume that steady-state conditions have been reached at the end of the analyzed period. Small head changes on the fringes of the mound are also difficult to measure and may have a significant impact because of the large area they cover.

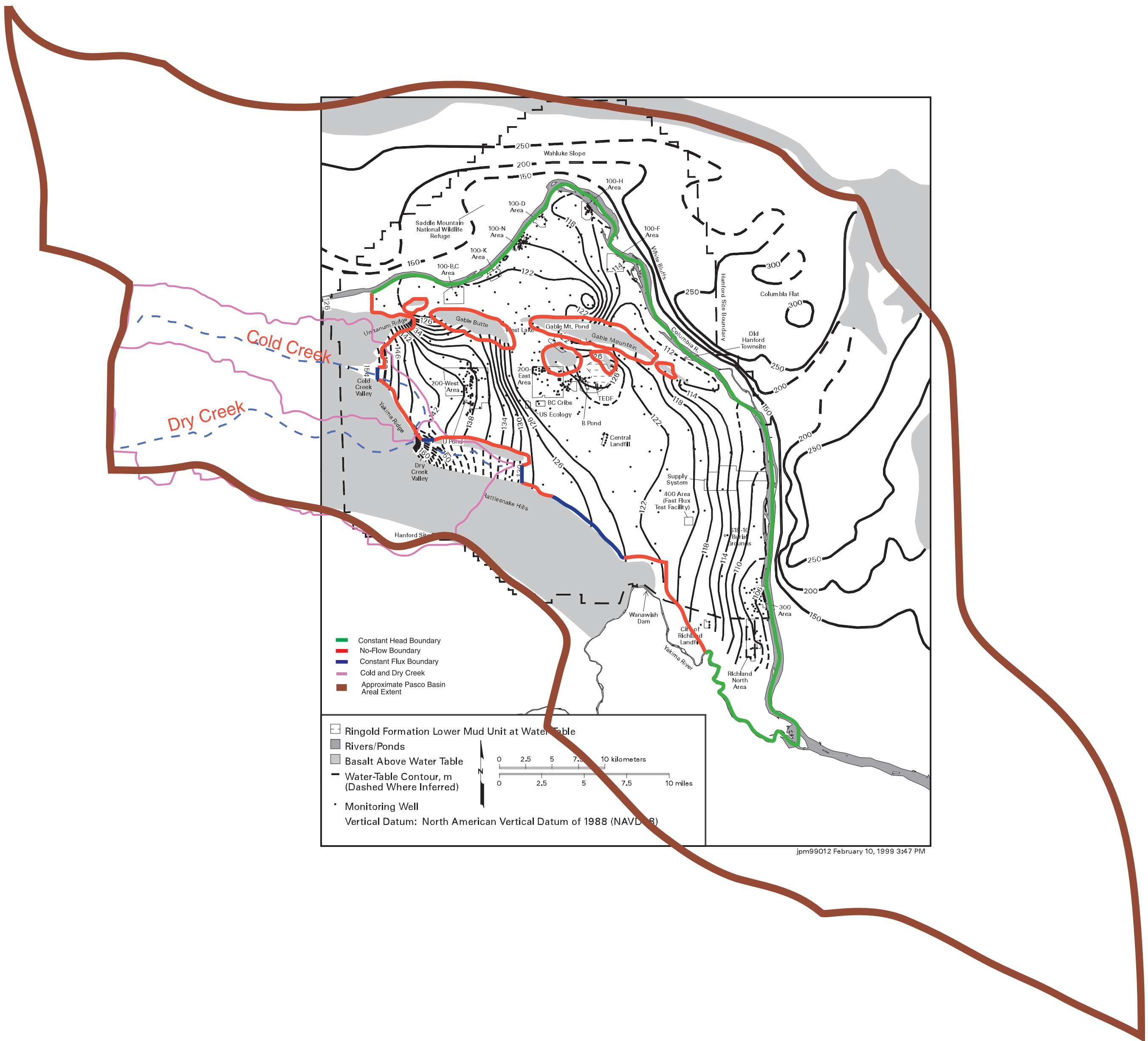
Specific yield for Unit 1 is estimated to range from about 0.1 to 0.3 and is expected to be higher for coarse, well-sorted gravel than for poorly sorted mixtures of sand and gravel. Storativity is estimated to range from 0.0001 to 0.0005. Specific yield is estimated to range from 0.05 to 0.2 for the generally poorly sorted sediments of Units 5, 7, and 9. Storativity is estimated to range from 0.0001 to 0.001 for these units.

D.2.1.6 Hydraulic Heads. Hydraulic head information is important to determine groundwater flow direction and velocity. Head measurements are also needed to establish initial conditions for groundwater flow modeling and for model calibration.

Water levels have been measured on at least an annual basis using a sitewide well network since the 1940s. More than 600 wells are currently measured each year to determine the hydraulic head distribution for the unconfined aquifer on the Hanford Site and adjacent areas. Results of the 1994 measurements are presented in Dresel et al. (1995). Additional water-level data for the North Richland area are provided in Liikala (1994). The annual water-level measurements provide an extensive database that can be used to define initial head conditions for numerical modeling and for a comparison of modeling runs with historical data. The interpreted watertable for the June 1998 water level data is provided in Figure D-17. Figure D-17 shows the groundwater level contour lines (i.e., lines of equal groundwater level) for the unconfined aquifer. Groundwater flow occurs at right angle to these contour lines, moving from higher to lower elevation. Locations where the basalt is above the water table are also shown. The basalt is assumed to be relatively impermeable, and the flowing groundwater must move around these obstructions to flow.

Before the mid-1980s, hydraulic heads increased by more than 13 m in some areas of the Hanford Site in response to wastewater disposal activities. Before wastewater disposal operations began, the uppermost aquifer was almost entirely within the Ringold Formation, and

Figure D-17. Hanford Site and Outlying Areas Water Table Map, June 1998.



the water table extended into the Hanford formation at only a few locations near the Columbia River (Newcomb and Strand 1953). However, wastewater discharges have caused the water-table elevation to rise into the Hanford formation near the 200 East Area and in a wider area near the Columbia River. Water levels have begun to decrease over most of the Hanford Site during the last several years because of decreases in wastewater discharge (Dresel et al. 1995).

Most of the wells in the current unconfined aquifer monitoring network are completed in the upper part of the aquifer, within 7 m of the water table. Most of the wells that were originally open to a greater depth interval were reconfigured in the early 1980s. The conceptual groundwater model, by its nature, is a 3-D problem and requires information on the vertical distribution of hydraulic head as well as the areal distribution. The locations and a listing of selected wells currently completed in the deeper part of the unconfined aquifer and wells with individual piezometers open to different depth intervals, is presented in Wurstner et al. (1995).

D.2.1.7 Transport Parameters. To accurately model contaminant transport using the advective-dispersion equation to describe spreading and linear isotherm process to describe contaminant velocity, parameters including effective porosity, dispersivity, and retardation coefficients must be specified. Longitudinal, transverse, and vertical dispersivity values are needed for a 3-D model. Retardation coefficients are specific to each contaminant species in association with the groundwater and host sediments. Thus, retardation coefficients may vary spatially and temporally depending on geochemical conditions within the aquifer. Information of retardation coefficients for Hanford Site unconfined aquifer sediments is available in Ames and Serne (1991) and Kaplan and Serne (1995).

D.2.1.7.1 Effective Porosity. Porosity is defined as the volume of void space divided by the total volume of the soil or rock matrix that contains it. Effective porosity does not include void space that is isolated from groundwater flow and, therefore, may be smaller than the total porosity. The average velocity of a conservative contaminant (non-sorbing and non-decaying), as it moves through an aquifer, is equal to the average linear velocity of the groundwater, which is inversely proportional to the effective porosity of the aquifer matrix (Freeze and Cherry 1979). Porosity can be determined from laboratory measurements on samples of aquifer material or from field tracer tests. For unconfined aquifers, effective porosity can be approximated by the same value used for specific yield, which is obtained from multiple-well hydraulic tests.

Laboratory measurements of porosity are available for samples from only a few of the available Hanford Site wells. Recently, 15 samples were collected from 6 wells at the 100-H Area (Vermeul et al. 1995). Porosity ranged from 0.19 to 0.41 and averaged 0.33 for the Ringold Formation and 0.31 for the Hanford formation. Samples from five depth intervals within the Ringold Formation at the 200 West Area were reported by Newcomer et al. (1995). The average porosity ranged from 0.21 to 0.33 and averaged 0.27. Laboratory porosity measurements are often considered unreliable, especially for unconsolidated sediments, because of the difficulty in obtaining undisturbed samples.

A few tracer tests have been conducted within the unconfined aquifer. Bierschenk (1959) reported an effective porosity of 0.10 from a tracer test with fluorescein dye under natural gradient conditions. Single borehole dilution tests, which do not provide information on porosity, were conducted by Graham et al. (1984). An effective porosity of 0.25 was assumed to calculate average groundwater velocity from the measurements. Borehole dilution tests and a two-well tracer test were conducted in the 200 West Area (Newcomer et al. 1995) under natural gradient conditions. However, porosity could not be determined from the two-well tracer test because the gradient was not well defined.

Porosity can also be estimated from measurements of aquifer specific yield. Specific yield is defined as the volume of water released from a unit area of an unconfined aquifer per unit decline in hydraulic head. Specific yield and effective porosity are equivalent if drainage of the aquifer matrix is complete. However, in reality, the specific yield may be lower than the effective porosity because of water held in pore spaces of the drained aquifer matrix by surface tension or adsorptive forces (Moench 1994).

As previously discussed, specific yield can be calculated from 1) multiple well aquifer tests; or 2) measurements of the volume of aquifer drained or saturated in response to removing or injecting a known volume of groundwater. The specific yield was calculated from the change in saturated aquifer volume associated with dissipation of the groundwater mound beneath the 200 West Area from 1985 to 1995. The result was a specific yield value of 0.17, which is higher than values calculated by Newcomb and Strand (1953) when they analyzed the growth of groundwater mounds beneath liquid disposal facilities in both the 200 West Area and 200 East Area between 1948 and 1953. Water levels beneath the 200-West Area had increased by an additional 5 to 10 m from 1953 to 1985. Therefore, the difference in porosity could be caused by a difference in the sediments saturated during the 1953 to 1985 period compared to those during 1985 to 1995. Specific yield results from the relatively few multiple well tests conducted on the Hanford Site unconfined aquifer range from 0.01 to 0.37 and average 0.15. However, some of these estimates are highly uncertain because the effects of nonideal test conditions (i.e., partially penetrating wells and aquifer heterogeneity). Such conditions generally have a more significant effect on the determination of storage properties than on determining transmissivity. Moench (1994) demonstrated that these conditions can affect specific yield values calculated from type-curve analysis of aquifer pumping tests, and usually result in the calculated values being low.

Mud-dominated units generally have higher porosity than sand and gravel-dominated units. Davis (1969) compiled porosity values that indicate ranges of 0.35 to 0.5 for silts and 0.4 to 0.7 for clays, respectively. However, because of the low permeability of such sediments, the porosity assigned to mud units in the model is not expected to have a major impact on model results.

D.2.1.7.2 Dispersivity. The following discussion on dispersivity, summarized from Kincaid et al. (1995), illustrates the factors that go into the selection of dispersivity values. Dispersivity is determined by inverse modeling of tracer test breakthrough curves from tests performed at the transport scale of interest and in the geohydrologic system of interest (Farmer 1986). Dispersivity has been called "the most elusive of the solute transport parameters" (Freeze and Cherry 1979) because it cannot be directly measured in the field or laboratory. Freeze and Cherry (1979) indicate that values of longitudinal and transverse dispersivities are significantly larger than values obtained in laboratory experiments on homogeneous materials and materials with simple heterogeneity. No field tests have been conducted at the Hanford Site to develop an estimate for this parameter at the scale of transport appropriate for the Site Groundwater Conceptual Model.

General studies indicate that dispersivity is a function of both time and transport distance because of unaccounted for temporal changes and unaccounted for heterogeneities. The U.S. Environmental Protection Agency (EPA), in their guidance for water quality assessment screening for toxic and conventional pollutants in surface and groundwater (Mills et al. 1985), indicates "A rough estimate of longitudinal dispersivity in saturated porous media may be made by setting D_l (cm) equal to 10% of the mean travel distance." This rule of thumb is based on analysis of tracer tests performed over a large range of laboratory and field scales and for a wide variety of aquifers.

The original work was performed by Lallemand-Barres and Peaudecerf (1978) and expanded by Gelhar and Axness (1981). Later in 1992, Gelhar et al. (1992) reexamined the data and indicated that because of the potential unreliability of the data that no definite conclusion regarding the rule could be reached beyond transport distances of 100 m. However, this rule was later refuted by Neuman (1993).

Dispersivity is theoretically expected to have an asymptotic value that can be related to the scale of uncharacterized aquifer heterogeneity (Farmer 1986). In contaminant transport simulations, large values of dispersivity result in lower peak concentration estimates, but give rise to earlier first arrival times that can increase arrival concentrations of radionuclides with short half-lives. Freeze and Cherry (1979) observed that longitudinal dispersivities, as large as 100 m and lateral dispersivities as large as 50 m, have been used in migration studies of large contaminant plumes. As discussed in Wurstner et al. (1995), the 1/10 approach has generally been used in the past to determine dispersivity values for Hanford Site transport modeling. Law (1992) used values of D_l = 43 m and D_t = 12 m for a scale of 9500 m based on values compiled in Gelhar et al. (1985). An earlier model (Golder Associates 1990) used values of 15 m and 1.5 m, respectively, for longitudinal and transverse dispersivity, which were also based on Gelhar et al. (1985).

It should be also recognized that the dispersivity values determined from field tests at 59 different sites compiled by Gelhar et al. (1992) included results from two investigations at the Hanford Site. The first was a 1950s tracer test that resulted in values of D_l = 6 m for the Hanford formation and D_l = 460 m for the Ringold Formation, as reported by Bierschenk (1959). Also included are values of D_l = 30.5 m and D_t = 18.3 m for a scale of 20,000 m. These were calculated from 2-D transport modeling of the 200 East Area tritium plume, as reported in Ahlstrom et al. (1977).

Dispersivity is likely to vary across the Hanford Site depending on the degree of heterogeneity and the temporal variability of flow gradients. Ahlstrom et al. (1977) noted that the ratio of D_l to D_t calculated from their model of the Hanford Site was much higher than the ratio expected. They attributed the high ratio to heterogeneity. However, horizontal dispersion may have been enhanced by temporal variations in flow gradients caused by disposal practices. The flow paths for the tritium transport from the 200 East Area have gradually shifted from due east to a south-easterly direction, in response to wastewater discharges to B Pond and the 200 East Area. This shift in the flow path has enhanced the apparent dispersion of the tritium plume emanating from the 200 East Area. More recent sitewide modeling studies (Law et al. 1996) used values of D_l and D_t of 30.5 m and 3 m, respectively, which appear to be related to the transport grid spacing of 100 m. In the recent *Hanford Low-level Tank Waste Interim Performance Assessment* (Mann et al. 1997) the horizontal dispersivity for aquifer transport was set at 10% of the travel length in the direction of flow and in the vertical direction at 1% of the travel length.

For the Composite Analysis (Kincaid et al. 1998), a longitudinal dispersivity, D_l, of 95 m was selected. While the value of D_l = 95 m is not based on any Hanford Site data, it satisfies all three of the following constraints on its value:

1. The numerical constraint is related to the grid Peclet number, P_e=(grid spacing)/D_l. For finite element transport simulations, P_e <4 are required for acceptable solutions (Campbell et al. 1981). The 95-m dispersivity estimate is approximately one quarter of the grid spacing in the finest part of the model grid in the 200 Area Plateau where the smallest grid spacing is on the order of about 375 by 375 m.

2. At the grid scale of 375 m used for the Composite Analysis modeling, the modeled system is homogeneous. Heterogeneities at scales less than 375 m are uncharacterized. The 95-m dispersivity value selected satisfies this constraint.

3. Finally, because it is more than 10 km from the closest source in the 200 East Area to the Columbia River, a nonasymptotic value of 1,000 m for the longitudinal dispersivity could be appropriate. Because large values of dispersivity are not conservative in transport simulations, the 95-m dispersivity value selected for use in the Composite Analysis is the smallest value that could be used with the grid spacing selected. Applying the rule of thumb, discussed previously, estimates of concentration 950 m from the source should be accurate and, for greater distances, they should be conservative.

With regard to transverse dispersivity the following is noted:

- EPA guidance (Mills et al. 1985) is 1/3 for the ratio of D_l / D_t.

Freeze and Cherry (1979) indicate transverse dispersivity is lower by a range of 5 to 20 (i.e., 0.2 to 0.05).

Walton (1985) states that reported ratios of D_l / D_t vary from 1 to 24, but that common values are 1/5 and 1/10.

As an example, the Composite Analysis in applying this guidance assumed the transverse dispersivity, D_t, was approximately 20% of the longitudinal dispersivity or about 20 m.

D.2.2 Alternative Representations to the Base Model

The base conceptual groundwater model presented in the previous section is the proposed integration of all the relevant features, processes, and events that are believed necessary to support the objectives of the initial SAC (Rev. 0). There are alternative representations to many of these features, processes, and events that should be considered for future efforts or perhaps for the initial effort. Provided in Table D-3 is a listing of potential alternative representations, the recommended representation for the base conceptual model, and justification for recommending the base representation.

Table D-3. Potential Alternative Representations to the Base Conceptual Model. (2 Pages)

Base Case Representation	Alternative Representation	Comment
Stratigraphically influenced saturated water flow and contaminant transport where migrating waste does not appreciably affect media or transport properties.	Stratigraphically influenced saturated water flow and contaminant transport, where migrating waste may appreciably affect either or both media and transport properties	Current contaminant plumes represent a large-scale in situ test that, when properly captured in a calibrated model, may automatically include the waste affects on media and transport for several CoCs (e.g., tritium, Tc-99, and nitrate). Additional laboratory and field data may be required for DNAPL contaminants.
Stratigraphically influenced saturated water flow and contaminant transport, where flow is stable.	Stratigraphically influenced saturated water flow and contaminant transport where flow is unstable in potential situations that may arise from groundwater fluid – wastewater fluid interfaces and differences in wetting properties such as may be associated with DNAPLs.	Current contaminant plumes represent a large-scale in situ test that, when properly captured in a calibrated model may automatically include the impacts of unstable flow for a large portion of the Site. Additional field data may be required.
Stratigraphically influenced saturated water flow and contaminant transport, where migrating waste is not appreciably affected by preferential flow.	Stratigraphically influenced saturated water flow and contaminant transport where migrating waste is appreciably affected by preferential flow.	Current contaminant plumes represent a large-scale in situ test that, when properly captured in a calibrated model, may automatically include the impacts of preferential flow for a large portion of the Hanford Site. Additional field data may be required.
Nine hyrdrostratigraphic layers are defined with variable finite thickness and homogeneous hydraulic and transport parameters.	Lump similar hyrdrostratigraphic units.	Groundwater flow paths are expected to vary from the nine layer conceptualization; it may help identify data needs.
Nine hyrdrostratigraphic layers are defined with variable finite thickness and homogeneous hydraulic and transport parameters.	Add additional layers to the nine layer base model.	The data are incomplete to support additional descritization. Extensive data collection and data reevaluation would be required.
Nine hyrdrostratigraphic layers are defined with variable finite thickness and homogeneous hydraulic and transport parameters.	Add discontinuities (e.g., sand lenses within a predominantly clay layer) to existing layers as may be inferred by the data.	Additional field data would be required to substantiate inferences.

Table D-3. Potential Alternative Representations to the Base Conceptual Model. (2 Pages)

Base Case Representation	Alternative Representation	Comment
Contaminant transport is assumed to spread per the standard dispersion equation with the velocity of groundwater for conservative contaminants or retarded per the linear isothermal process.	Revise contaminant transport to include reversible reactions.	There are very little laboratory data and no site-specific field data to support inclusion of the reactive transport process.
Contaminant transport is assumed to spread per the standard dispersion equation with the velocity of groundwater for conservative contaminants or retarded per the linear isothermal process.	Revise contaminant transport to include multiphase density saturated flow for DNAPLs.	Additional field data would be required.
Hydraulic conductivity is assumed isotropic in the horizontal plane (i.e., Kx = Ky) and one tenth the horizontal value in the vertical direction (i.e., 0.1Kx = Kz).	Include variable anisotropy of hydraulic conductivity.	Extensive data collection would be required.

D.2.3 Uncertainty in the Conceptual Model

There are two major areas from which uncertainty in the conceptual groundwater model arise:

- Model uncertainty or the chance that the conceptual groundwater model is inappropriate (e.g., has assumed inappropriate processes such as linear sorption isotherm process for transport, or that there is no communication with the underlying basalts) for the objectives and approach of the SAC (Rev. 0).

- Parameter uncertainty associated with prescribed processes, hydrogeologic features, boundary conditions, stresses imposed on the system, hydraulic parameters, and transport parameters, which are not known everywhere in the model domain.

Model uncertainty can be estimated with select test cases. The process involves 1) identifying alternative conceptual groundwater models; 2) establishing a set of test cases where the metric would include such things as time versus concentration of a contaminant(s) at selected points in the domain; 3) expressing the alternative conceptual models numerically; 4) calculate the appropriate metric values; and 5) then compare results. The alternative conceptual models would initially be developed from combinations of alternative representations provided in Table D-3. The process would likely be iterative, as results were obtained and additional combinations of alternative realizations were considered.

Parameter uncertainty is the result of the difficulties associated with quantifying the nature and degree of spatial and temporal variations in groundwater flow and contaminant transport parameters along the entire pathway(s). Parameter uncertainty is also extended to the processes, boundary conditions, and stresses imposed on the system. Parameter uncertainty can be estimated with sensitivity analysis and uncertainty analyses.

Sensitivity analysis is a quantitative method of determining the effect of parameter variation on model results. A sensitivity analysis quantifies the uncertainty in the calibrated model caused by uncertainty in the estimates of aquifer parameters, stresses, boundary conditions, and processes. The analysis identifies the model inputs that have the most influence on model calibration and predictions, can provide an understanding of the level of confidence in model results, and identifies data deficiencies. The process involves 1) selecting the calibrated model that best represents the hydrogeologic system (for the purposes and approach identified for the SAC (Rev. 0); 2) establishing a set of parameter ranges; 3) establishing a set of test cases where the metric would include such things as time versus concentration of a contaminant(s) at selected points in the domain; 4) calculating the appropriate metric values; and 5) comparing results qualitatively and statistically.

D.2.4 Assumptions and Rationale

Table D-4 lists the major assumptions that are incorporated into the proposed base model.

Table D-4. Major Assumptions Incorporated into the Base Conceptual Model. (2 Pages)

Assumption	Rationale	Impact
The conceptual groundwater model is not considered to be stochastic.	The Hanford Site has received extensive evaluation and study of a 50-year period. Past contaminant release afford the opportunity to monitor plume migration.	There is inherent uncertainty associated with one conceptual model. The uncertainty can only be estimated with testing and comparison with alternative conceptual models.
The Columbia River is treated as a constant head boundary with stage elevation in the river based on time-averaged data.	Bank storage is ignored, and flow variations near the river are assumed to be insignificant for the long-term problems (i.e., 1,000 years) that the SAC (Rev 0.) will address	Short-term variations (i.e., daily, weekly) will not be captured.
There is no aquifer communication between the unconfined aquifer and the confined aquifer.	The hydraulic conductivity of the rock separating the two-aquifer systems is low. There are some localized areas of intercommunication (i.e., the Gable Mountain and Gable Butte area), based on chemistry data and near the Yakima River horn where the river has incised the upper basalt confining layers.	Contaminant concentration predictions for the n- flow case would be higher than for the case with potential inflow to the unconfined aquifer, because inflow would result in dilution. For flow to the confined aquifer, a segment of a pathway is ignored. However, the confined aquifer likely discharges back into the unconfined aquifer.

Table D-4. Major Assumptions Incorporated into the Base Conceptual Model. (2 Pages)

Assumption	Rationale	Impact
Contaminant flow and transport in the unconfined aquifer system is adequately represented by nine hydrostratigraphic units.	Definition (i.e., thickness, extent, and parameter values) of the nine units are supported by available data and represent all major and areally extensive conductive and nonconductive geohydrologic units above the basalt.	Captures potential pathways not previously captured with models of fewer units. Greater computational demand potentially required when compared to conceptual model with fewer units. Data are sparce in several areas resulting in uncertainties about lateral continuity and parameter values. Additional units may better represent potential flow paths; however, extensive field and laboratory data would be required to support development.
Natural recharge is variable across the Hanford Site and is included as a surface condition.	Variability of recharge across the Hanford Site is based on the distribution of surface cover, ranging from natural shrub-steppe vegetation in a fine-grained soil to bare gravel surfaces. The differences in recharge, based on surface cover, have been well documented (Fayer and Walters 1995).	The variation of recharge across the Hanford Site affects the flow model calibration. The result is a distribution of higher hydraulic conductivity than would occur without recharge. Recharge affects the contaminant transport results by diluting the contaminant concentrations and driving the maximum concentrations to below the water table surface.
Contaminant fate and transport is adequately represented by the linear sorption isotherm process.	It is the only transport process for which Hanford Site data are available.	Spatial and temporal variations in the transport of contaminants may not be captured because of reactions (e.g., chemical and biotransformation) that occur along the pathway.
Parameters are generally considered to be isotropic in the horizontal plane (e.g., $K_x = K_y$).	Although there are hydrogeologic heterogeneities, few if any tests have been designed to quantify anisotropy associated with the heterogeneity.	The adequate representation of contaminant plume spreading becomes more dependent on the assumed dispersivity values, and additional uncertainty may arise from difficulties associated with estimating dispersivity.
Preferential flow is not considered explicitly.	Variation of hydraulic conductivity assumed in the conceptual model may result in some preferential flow.	Potential occurrences of earlier first arrival, higher contaminant concentrations, and the arrival of other less mobile contaminants will not be captured.

D.3 OUTSTANDING ISSUES

The major outstanding issues have been documented in Gorelick et al. (1999). An extract from their executive summary describes these issues and is provided in the following.

With regard to improvements in the modeling framework:

- The existing deterministic modeling effort has not acknowledged that the prescribed processes, physical features, initial and boundary conditions, system stresses, field data, and model parameter values are not known and cannot be known with certainty. Consequently, predictions of heads and concentrations in three dimensions over time will be uncertain as well.

- A new modeling framework must be established that accepts the inherent uncertainty in model conceptual representations, inputs, and outputs. Given such a framework, the expected values of heads and concentrations, as well as the range (distribution) of predictions, would be products of the sitewide groundwater model.

- A priority task is to construct a comprehensive list of alternate conceptual model components and to assess each of their potential impacts on predictive uncertainty.

- Assessment can be initiated with hypothesis testing and sensitivity analysis within the general framework already established with the existing sitewide model. If uncertainties due to alternate conceptual models are significant, then a Monte Carlo analysis is required to estimate both the expected value of the prediction and its uncertainty.

D.4 PROPOSED PATH FORWARD

The proposed path forward addresses the two major areas from which uncertainty in the conceptual groundwater model arise: model uncertainty and parameter uncertainty. In general technical terms, the proposed path forward has already been described in Section D.3. The implementation should address the open issues identified by the peer review panel (Gorelick et al. 1999), as summarized in Section D.3. Additional field and laboratory data may be required but this need is not certain until the analyses discussed in Section 1.3.3 have been completed.

Success of this proposed path forward hinges on the ability to elicit and act upon input from Ecology, the Indian Nations, and other interested stakeholders. The approach is iterative. Additional alternative representations may be developed based on the initial results and additional data, if collected, would need to be incorporated and the results, in the form of a revised or refined conceptual groundwater model evaluated.

D.5 REFERENCES

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